Emerging Contaminants and Associated Treatment Technologies

Tariq Aftab *Editor*

Emerging Contaminants and Plants

Interactions, Adaptations and Remediation Technologies

Springer

Emerging Contaminants and Associated Treatment Technologies

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Editor
Tariq Aftab
Department of Botany
Aligarh Muslim University
Aligarh, Uttar Pradesh, India

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Preface

The “emerging contaminants” (ECs) are predominantly unregulated anthropogenic chemicals that occur in air, soil, water, food, and plant/animal tissues in trace concentrations. The ECs are persistent in the environment, capable of perturbing the physiology of target receptors and, therefore, are regarded as contaminants of emerging environmental concerns in recent years. The prominent classes of ECs include pharmaceuticals and personal care products (PCPs), surfactants, microplastics, pesticides, fire retardants, and nanomaterials. If released into the environment, these contaminants may enter the food chain. Emerging contaminants influence the physiological cell reactions at different levels and lead to altering in normal cell function primarily at the molecular and biochemical level. Molecular responses to such ECs have been studied over the last few years, in which there is an intricate network of signaling pathways controlling perception of these environmental stress signals, the generation of second messengers, and signal transduction. Recent advances in many areas of plant and microbial research, including genotyping, make scientists optimistic that valuable solutions will be found to allow deployment/commercialization of strategies better able to tolerate these ECs.

ECs may also be created in the agricultural environment itself. Many man-made substances that enter agricultural systems will be degraded by chemical, physical, and biological processes. While these reactions may result in the complete breakdown of a chemical (i.e., they convert the chemical to carbon dioxide, water, and methane), in many instances the processes may result in the formation of stable intermediate chemicals which we call degradation or transformation products. Many ECs are also produced naturally by fungi, bacteria, algae, plants, and animals. According to their origin, these toxins are usually separated into the categories: mycotoxins (fungal origin), bacterial toxins (bacterial origin), phycotoxins (algal origin), phytotoxins (plant origin), and zootoxins (animal origin). These are usually produced by the organisms as a defense mechanism. The ECs where agriculture is currently the predominant source of surface water contamination, compared to non-agricultural sources, are the natural toxins, veterinary medicines, hormones, and transformation products of man-made chemicals used in agriculture. For some ECs, agriculture might be an important source of contamination in the future. For example, if pesticides are developed in the nano-form in the future, it is likely that these materials will be transported to and contaminate surface waters. For some ECs, for example, bioterrorism/sabotage agents, the importance of agriculture as a source of contamination is difficult to predict.

Environmental remediation was historically viewed as an inherently sustainable activity, as it restores contamination; however, researchers and practitioners are increasingly recognizing that there can be substantial environmental footprints and socioeconomic costs associated with remediation. Sustainability is an imperative in the emerging green and sustainable remediation movement, which is reshaping the entire remediation industry. The book comprises diverse chapters accumulating the significant roles of ECs in affecting plants and environment, and the need of sustainable or eco-friendly approaches in mitigating these ECs

Aligarh, Uttar Pradesh, India Tariq Aftab

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About the Editor

Tariq Aftab received his PhD from the Department of Botany at Aligarh Muslim University, India, and is currently an assistant professor there. He is the recipient of a prestigious Leibniz-DAAD fellowship from Germany, Raman Fellowship from the Government of India, and Young Scientist Awards from the State Government of Uttar Pradesh (India) and Government of India. After completing his doctorate, he worked as a research fellow at the National Bureau of Plant Genetic Resources, New Delhi, and as a postdoctoral fellow at Jamia Hamdard, New Delhi, India. Dr. Aftab also worked as a visiting scientist at Leibniz Institute of Plant Genetics and Crop Plant Research (IPK), Gatersleben, Germany, and in the Department of Plant Biology, Michigan State University, USA. He is a member of various scientific associations in India and abroad.Tariq has edited 15 books with international publishers, including Elsevier Inc., Springer Nature, and CRC Press (Taylor & Francis Group), co-authored several book chapters, and published more than 80 research papers in peer-reviewed international journals. His research interests include physiological, proteomic, and molecular studies on medicinal and crop plants .

Chapter 1
An Insight Into the Consequences of Emerging Contaminants in Soil and Water and Plant Responses

Masina Sairam, Sagar Maitra, Subhashisa Praharaj, Suprava Nath, Tanmoy Shankar, Upasana Sahoo, D. T. Santosh, Lalichetti Sagar, Monalisha Panda, G. Shanthi Priya, T. R. Ashwini, Dinkar J. Gaikwad, Akbar Hossain, Biswajit Pramanick, Hanuman Singh Jatav, Harun I. Gitari, and Tariq Aftab

Abstract With the advancement of science, better monitoring of soil and water quality has become possible. Many contaminants have been reported in the recent past that influence the quality of soil and water negatively. However, the consideration of these pollutants or contaminants is still in the initial stage and needs to be explored in detail for a better understanding of their activity as contaminants.

M. Sairam · S. Maitra (✉) · T. Shankar · U. Sahoo · L. Sagar · M. Panda · G. Shanthi Priya
T. R. Ashwini · D. J. Gaikwad
M.S. Swaminathan School of Agriculture, Centurion University of Agriculture and Management, Paralakhemundi, Odisha, India
e-mail: sagar.maitra@cutm.ac.in

S. Praharaj · B. Pramanick
Dr. Rajendra Prasad Central Agricultural University, Pusa, Samastipur, Bihar, India

S. Nath
Department of Agronomy, University of Agricultural Sciences, Bangalore, Karnataka, India

D. T. Santosh
School of Agriculture and Bio-engineering, Centurion University of Agriculture and Management, Paralakhemundi, Odisha, India

A. Hossain
Bangladesh Wheat and Maize Research Institute, Dinajpur, Bangladesh

H. S. Jatav
Sri Karan Narendra Agriculture University, Jobaner, Jaipur, Rajasthan, India

H. I. Gitari
School of Agriculture and Enterprise Development, Kenyatta University, Nairobi, Kenya

T. Aftab
Department of Botany, Aligarh Muslim University, Aligarh, Uttar Pradesh, India

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Emerging contaminants such as agrochemicals, nanomaterials, pharmaceuticals, personal care products, and micro- or nanoplastics have been found to show several harmful impacts on soil or water quality. Emerging contaminants are known to have adverse effects on plants and human beings too. The risk of their entry into the crops, food chain, and any possible interaction to human health should be properly monitored. The concentration of these contaminants in soil and water should also be monitored on a regular basis to avoid the significant damages arising from them. Future study may also be taken into consideration to avoid the possible concerns to natural resources, plants, and human wellbeing.

Keywords Microplastic · Nanomaterials · Pharmaceuticals · Agrochemicals · Personal health care products · Plants

1.1 Introduction

Soil and water are the basic natural resources for agriculture. Soil and water quality plays an important role in deciding farm productivity and agricultural sustainability. Rapid industrialization, urbanization, and increased human population lead to generation of a large volume of wastes which are mostly accumulated in soil/water leading to soil and water contamination. Accumulation of contaminants in soil causes reduction in the productive potential of soil, poor microbial activity, and overall reduction in crop productivity. Knowing about the status of pollutants in soil, their nature, and possible negative impacts is very crucial for understanding the usability of soil and water for particular use. Considering the fact that rising population and rapid urbanization will lead to lesser availability of land and water for agricultural use, it is vital to sustain or improve soil and water quality in all aspects for a healthy growth of crops.

During recent times, an enhanced awareness and concern about the emerging pollutants in soil and water has been noted. Even though their concentration is of very low concentration, they are highly toxic and have significant impact on soil or water quality. Most of the pollutants irrespective of their origin often end up in soil as well as water systems; hence, the risk of soil or water contamination is very high especially in regions with high industrialization. Human activities such as disposal of industrial waste, mining, and excessive and indiscriminate use of agrochemicals in agriculture are few of the contributors to emerging contaminants in soil. Even though the emerging contaminants are highly toxic, very little attention has been given to these emerging contaminants until recent periods (Petousi et al., 2015; Baybil et al., 2022). The risk of entry of these contaminants into the food chain and their possible biomagnification cannot be ignored. In this chapter, an attempt has been made to discuss the major emerging contaminants in soil and water and their adverse impacts on the environment and lifeforms (Fig. 1.1).

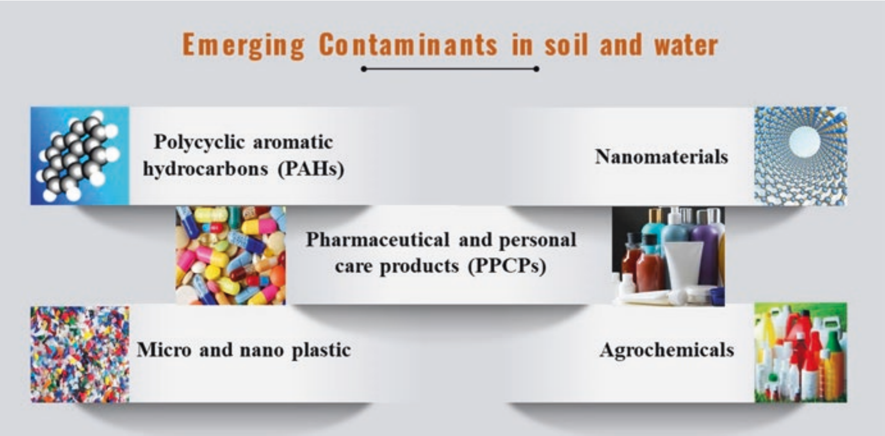

Fig. 1.1 Different types of emerging contaminants in soil and water

1.2 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are organic compounds originated naturally in fossil fuels, chemically composed of carbon and hydrogen atoms with benzene rings, colourless or pale yellow coloured, a group of numerous chemically related compounds, and toxic on living organisms (Abdel-Shafy & Mansour, 2016; Drotikova et al., 2021). PAHs are known to suppress immunity and develop mutagenic and carcinogenic effects (Armstrong et al., 2004). In general, PAHs are grouped into three categories, namely, pyrogenic, petrogenic, and biological. In the pyrolysis process, when organic materials are exposed to heat with low or no oxygen conditions, the pyrogenic PAHs are produced. Further, incomplete combustion of motor fuel, forest fire, and burning of cigarettes also produces pyrogenic PAHs (Baklanov et al., 2007). The petrogenic process occurs during crude oil maturation. Biologically PHAs are produced in bacterial and algal synthesis, decomposition of vegetable oil, etc. (Abdel-Shafy & Mansour, 2016). PAHs enter into the environment in several ways. They are found in the air, soil, and water. Like other contaminants, soil PAHs also tend to accumulate in soil. The lipophilic and hydrophobic nature of PAHs makes them persistent resulting in their long-term presence in soil (Li et al., 2019). The persistence of PAHs has also been linked to their number of rings. PAHs with higher rings have been found to be more persistent than PAHs with lower rings (Johnsen et al., 2005). PAHs with a lesser number of rings get solubilized easily in water which make them easily decomposable (Baybil et al., 2022). They have UV absorbance spectra, and each ring has a unique UV spectrum (Abdel-Shafy & Mansour, 2016). PAHs have high melting and boiling points with less solubility in aqueous solution (Masih et al., 2010). Commonly, different PAHs are used in industry in making of colours and dyes, agrochemicals, pharmaceuticals, and plastics, and the use of these products is continuously increasing (Fig. 1.2). PAHs

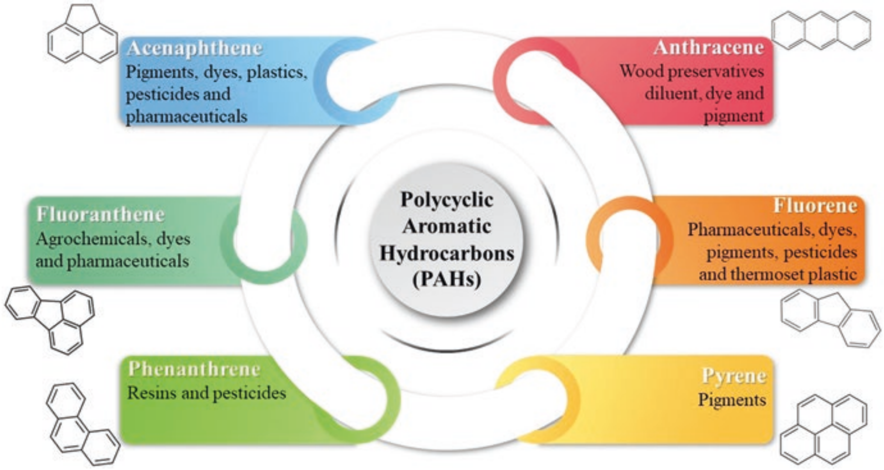

Fig. 1.2 Some common PAHs and the manufactured products

exposed to the atmosphere are deposited to the soil from fuel combustion. Some agricultural inputs such as pesticides and fertilizers may contain PAHs which are applied to the soil, and they become bound to soil particles. The sorption process governs the movement of bound PAH into the soil particle (Abdel-Shafy & Mansour, 2016). PAHs exposed to the atmosphere can be deposited to the natural waterbody, dispersed with the water flow, and settled as sediments (Sany et al., 2012). The PAHs with low molecular weight are soluble and can be dissolved into water.

PAHs show toxicity to aquatic flora and fauna by affecting their metabolic processes. Besides, birds and invertebrates are also adversely affected by PAHs. Mammals including human beings are exposed to PAHs mainly by inhalation, ingestion, and dermal contact (Dong et al., 2012; Veltman et al., 2012; Honda & Suzuki, 2020). Plants absorb PAHs present in the polluted soil, and thus, these enter into the food chain. However, they rarely show phytotoxicity unless concentration is more. The impact of PAHs on human health and plant health has been recorded in the earlier studies (Jameson, 2019; Alkio et al., 2005). The adverse health impact of PAHs on the human body may be acute (ACGIH, 2005; Unwin et al., 2006) and chronic (Diggs et al., 2011; Olsson et al., 2010). Eye irritation, vomiting, skin disease, diarrhoea, nausea, allergy, and confusion are some common health issues that can be caused because of exposure to PAHs. However, reduced immunity, eye disease, jaundice, liver and kidney damage, breathing problems, lung infection, etc., are the chronic health problems. Carcinogenic effect of PAHs has been studied (IARC, 2010). Contrasting results have been reported on the impacts of PAHs on plant growth processes (Baybil et al., 2022). Adverse effects of PAHs are also demonstrated in earlier studies (Alkio et al., 2005; Liu et al., 2009a, b).

PAHs are removed from the environment mainly by biodegradation and photochemical degradation (Sultana et al., 2021). The anaerobic degradation method is adopted in biodegradation (Schraa, 1988). The dissolved or vapour form of PAHs is bioavailable. PAHs sorbet into soil is not easily degraded by microorganisms as PAHs are parted from the enzymes that are used by the bacteria to break them down (Al-Turki, 2009). The PAHs that bonded with soil particles for a longer time show slow desorption. Phenanthrene and chrysene with angular carbon rings are the PAHs that biodegraded faster than others (Wang et al., 2009). The aerobic and anaerobic bacterial species, fungi, extremophiles, archaea, and algae are the bioagents used in microbial degradation (Patel et al., 2020).

Photolysis is another method of degradation wherein the presence of light PAHs stimulates the electrons by making unstable structural arrangements. The structure of PAHs is the determinant of photolysis. Angular structured PAHs degrade slowly; however, photolysis is more effective on the compounds with low molecular weight (Xu et al., 2022). Chemical oxidation is another method of PAH degradation. The effectiveness of chemical oxidation depends on the molecular structure and weight of the compound, oxidizing agents used, and temperature conditions. Titanium oxide and zinc oxide are the catalysts that facilitate UV irradiation of PAHs (Zhang et al., 2008). High-frequency ultrasound is also effective in the degradation process. From the atmosphere also, PAHs are removed by dry deposition (Manariotis et al., 2011) and wet deposition (Wang et al., 2010) methods.

1.3 Nanomaterials

Nanomaterials (NMs) are the materials having particles or constituents of nanoscale dimension, i.e. 1–100 nm (US EPA, 2007), which is produced through nanotechnology. Based on their origin, NMs are broadly categorized into three types such as natural, incidental, and engineered nanomaterials (US EPA, 2008). Most of the NMs that are used in different sectors inclusive of farm inputs are of engineered type and commonly known as engineered nanomaterials (ENMs). ENMs can be made through major three processes such as physical, chemical, and biological (Kolahalam et al., 2019; Baig et al., 2021; Varghese et al., 2019). NMs exhibit special properties which are different from their original forms in terms of physical, chemical, and mechanical strength and electrical conductivity (Bhushan et al., 2014; Lue, 2007; Wu et al., 2020; Saleh, 2020). Properties of soil as well as physicochemical characteristics of ENMs determine the effect of NMs on plants (Nowack et al., 2012). ENMs are widely used in various sectors such as manufacturing of semiconductor, cosmetics industry (Raj et al., 2012; Fytianos et al., 2020), medical imaging (Wen & Steinmetz, 2016), drug delivery (Alizadeh & Salimi, 2021), agricultural input (Peters et al., 2016; Kaphle et al., 2018; Durgude et al., 2022), food sector (Hossain et al., 2021), pollution sensors, and environmental remediation due to their unique properties which make them more efficient. However, environmental impacts and safety standards have been relatively less known.

Manufactured NMs are exposed into the environment through intentional and unintentional releases. They are found in atmospheric emissions, wastes from production facilities, paints, fabrics, agro-inputs, and personal care products (PCPs).

The NMs used eventually mix into the soil and water, and thus they contaminate the valuable natural resources (Ray et al., 2009). Application of biosolids and wastewater is one of the potential sources of addition of nanomaterials in agriculture soils (Pan & Xing, 2012) other than use of nano-inputs to the crop fields, plants, and foods (Hossain et al., 2021; Durgude et al., 2022). NMs are also released to the environment through faulty practices of waste disposal of some products such as cosmetics and electronic devices (Musee, 2011), and ultimately, they are transported to water bodies by runoff (Klaine et al., 2008). The transport of nanoparticles varies largely based on their size, surface chemistry, and various processes in the environment (USEPA, 2010). NMs released to the environment may react with natural materials and remain suspended individually or in aggregate. NMs can persist in air and water as they are in nano-size. The NMs in the porous media become attached to the mineral surfaces (Wiesner et al., 2006). Earlier studies reported that natural NMs that are existing in the nature from the very beginning of the Earth's history (Handy et al., 2008) show a negligible impact on environment and agroecosystem (Rohila et al., 2017); however, the ENMs which are in various use including agricultural inputs can be fatal by causing risks to agroecosystem or humans (Belal & El-Ramady, 2016), and ENMs are of prime importance as emerging contaminants. On the other hand, NMs are used for amelioration of problematic soils and polluted water (Stuart & Compton, 2015; Gil-Díaz et al., 2016). The adverse impacts of NMs are actually the adverse impacts of ENMs into the nature which are narrated in the following paragraphs.

ENMs are applied to the soil and environment for some purposes, and these materials are metal salts (ceramics and nano-silicates); metal oxides (nano-zinc oxide, nano-titanium oxides); nanoparticles of iron, gold, and silver; and carbon nanotubes (Belal & El-Ramady, 2016). Sometimes, unintentional combustion of products may also release NMs to the natural environment (Zhang et al., 2015). NMs are dispersed into the soil and make changes in soil particle aggregates, suspension ability, bioavailability, and transport (de Santiago-Martín et al., 2015). The suspended NMs can alter the physico-chemical properties including ionic strength as well as organic matter content (Delay et al., 2015; Li et al., 2016). They may also develop toxicity (Grillo et al., 2015; Li et al., 2016). NMs may form biofilms by mixing with dissolved organic matter in the presence of root exudates and secretion of microbes (Delay et al., 2015; Hüffer et al., 2018), and the carriage of essential macromolecules is altered (Belal & El-Ramady, 2016). Earlier studies reported that nanoparticles reduced growth of wheat (Dimkpa et al., 2013), nano-ZnO altered transportation mechanism in plants (Lin & Xing, 2008), of nano-Ag registered the cytotoxic and genotoxic impacts (Kumari et al., 2009) and changed the genetic makeup (Prakash et al., 2014; Frazier et al., 2014).

However, NMs are complex in nature, and their interaction in soil is also complex (Pan & Xing, 2012). Therefore, the assessment of risks while applying NMs is to be considered based on the nature and properties of soil and NMs. Soil microorganisms influence the soil biological properties, and they are important in influencing soil fertility and productivity as well as agroecosystem (Maitra et al., 2021). There are several NMs that show anti-microbial properties, and certainly they adversely affect the microbial population (Shah & Belozerova, 2009; Kumari et al., 2014; Van Aken, 2015; Sirbu et al., 2016). In general, NMs have the potential for amelioration or promotion of microbial toxicity (Thul et al., 2013). Based on the studies conducted so far on this aspect, it may be stated that NMs have enough potential to impact on the soil and microbe system and waterbodies; however, further and detailed investigation is needed.

Nanomaterials can pass through the placenta as well as the blood brain barrier owing to their smaller sizes (Liu et al., 2009a, b). Circulatory system of human beings gets affected due to continuous exposure of various nanomaterials (SCENIHR, 2009). The use of sunscreens (which contains mostly nano-TiO_2 and nano-ZnO) exposes the skin to nanomaterials, and these nanomaterials penetrate human skin (Mortensen et al., 2008). Drinking water contaminated with nanomaterials will cause ingestion exposure. Carbon and silica NMs may cause harmful diseases such as pulmonary inflammation, fibrosis, and granulomas, whereas nanoparticles of MnO_2 and TiO_2 adversely affect the brain (Ray et al., 2009).

1.4 Pharmaceutical and Personal Care Products

Pharmaceuticals are therapeutic drugs or medicines that cure and prevent ailments of human and animals, and these are prescribed by registered practitioners (Ebele et al., 2017). However, personal care products (PCPs) are products such as toothpaste, cosmetics, perfume and deodorants, and skincare items (Boxall et al., 2012; Baybil et al., 2022). These two categories of products are collectively known as pharmaceuticals and personal care products (PPCPs) and considered as emerging pollutants of the environment, especially water systems. In the process of manufacture, application, and disposal, the active pharmaceutical ingredients (APIs) are exposed to the environment including river water (Wilkinson et al., 2022) disturbing the aquatic ecosystem. During recent times, the PPCPs have become one of the major threats for human and livestock populations. The European Union (EU) and the US Environmental Protection Agency (USEPA) together listed more than 30 emerging pharmaceutical products exposed to water bodies causing pollution (Ebele et al., 2017). During the period of COVID-19 and afterwards, usage of facemasks has increased, and disposal of used masks further created a worldwide problem. Moreover, PPE kit, surgical face masks, gloves, and gowns are other accessories used in medical treatment having enough potential to pollute the environment (Fig. 1.3). The diversified nature of PPCPs, continuous introduction of new formulation, and increasing use are further adding dimension in this regard.

The conventional procedures of water treatment have some limitations as they cannot remove the physical and chemical properties of PPCPs, and they may persist even in the treated drinking water (Snyder, 2008) imposing the possibility of risks to public health. The pharmaceuticals creating environmental hazards are broadly categorized as persistent and pseudo-persistent (Ebele et al., 2017). The PPCPs when remaining biologically active in waterbodies can influence the normal

Fig. 1.3 PPCPs as emerging pollutants

biological activities of aquatic animals (Mimeault et al., 2005). The pharmaceuticals containing polychlorinated biphenyls, perfluoroalkyl substances, and polybrominated diphenyl ethers even in low concentration have the potential to alter some metabolic and enzymatic activities of aquatic animals (Fabbri & Franzellitti, 2016). Further, these may influence endocrine activities of aquatic lifeforms causing disruption of homeostasis leading to adverse health impacts. Various sex hormones, veterinary hormones, glucocorticoids, and some other non-hormone pharmaceuticals and PCPs show endocrine disruption (Wielogorska et al., 2015). The endocrine disruptors (EDs) consist of chemicals of natural or synthetic products and various PCPs. Moreover, the mixed form of different EDs with various concentrations can show synergy and fatal effects. The PPCPs are potentially harmful for development of antibiotic-resistant bacterial strains (WHO, 2015). The use of antibiotic is increasing against human and animal diseases, and when a portion of the antibiotic is mixed into the soil and waterbody, it can cause detrimental effect on microbes. It may be caused by development of antibiotic resistance in harmful bacteria and harmful impact on beneficial bacteria in the environment. Additionally, the active ingredients present in PPCP in the waterbody may change their toxicological properties (Isidori et al., 2005). Some acidic medicines when exposed to basic water may exhibit toxicity to non-target organisms (Fent et al., 2006). Actually, the toxicity due to PPCPs greatly depends on the tolerance of the exposed organisms, duration of exposure, and concentration of the harmful ingredient.

The recent addition in this category is mainly because of COVID-19 incidence and post-pandemic situation. This pandemic disease took thousands of lives in the world, and still it is creating problems in a scattered manner in different corners. COVID-19 has taught us to wear face masks and personal protective equipment (PPE) kit. Moreover, surgical accessories such as face masks, hand gloves, and gowns are dispensed after single use as these are non-reusable. They are made up of polypropylene, polyurethane, polyacrylonitrile, polyethylene, and polyethylene terephthalate that take several years to decompose (Fadare & Okoffo, 2020). Globally, 129 billion face masks and 65 billion gloves were used per month during the COVID-19 pandemic (Prata et al., 2020; Zhang et al., 2021). Microplastics and nanoplastics are the materials used in making face mask, causing a threat to the environment. COVID-19 is an infectious disease, and so the used facemasks were not recycled and landfilled (Shirvanimoghaddam et al., 2022); however, proper sterilization prior to landfilling was not followed in many cases. Under environmental conditions, facemasks can undergo different changes because of temperature, UV radiation, or a change in pH (Abbasi et al., 2020) and pollute soil and water. The microplastics can easily enter into the human respiratory system and gastrointestinal tract causing health hazards (Ragusa et al., 2021).

The PPCPs pollute the environment not only in the manufacturing units (Foltz et al., 2014) but also in post-use stages, and they enter the water and soil through different routes (Fig. 1.4). Sewage treatment plants (STPs), wastewater treatment plants (WWTP), and landfill leaching are the most prominent routes (Daughton & Ternes, 1999). The ineffectiveness of traditional wastewater treatment plants in removing these products results in their exposure into the soil and water (Bolong et al., 2009; Cabeza et al., 2012; Baybil et al., 2022).

Incomplete decontamination especially that of pharmaceuticals during treatment process has been attributed as the major reason for accumulation of these emerging contaminants in aquatic bodies where these treated wastewaters are disposed (Cabeza et al., 2012; Baybil et al., 2022; Petrie et al., 2015). Wastewater treatment plants generate huge amounts of sewage sludge (Praharaj et al., 2022; Sagar et al., 2022). Sewage sludge is used in agriculture as it contains various nutrients such as N, K, and Fe (Yang & Toor, 2015). However, application sewage sludge also leads to inadvertent addition of many toxic contaminants into soil. Treated sewage sludge

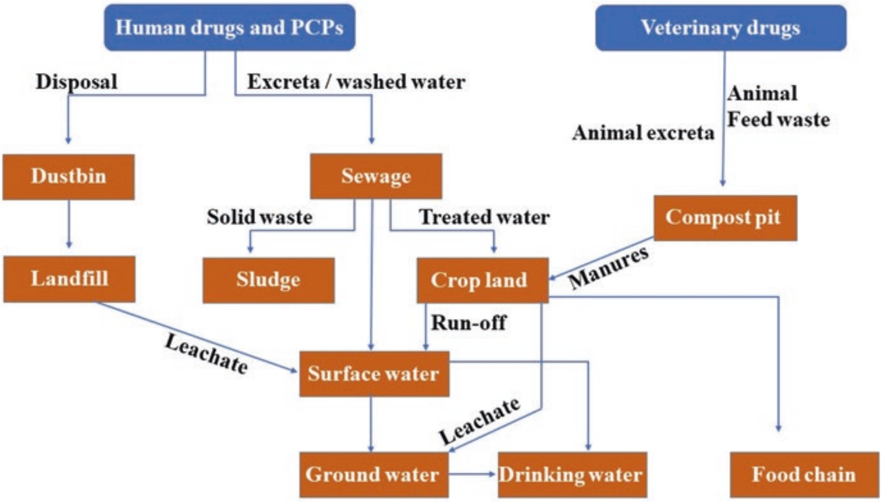

Fig. 1.4 Route of exposure of PPCPs to soil and water

contains many emerging pollutants, namely, antibiotics, chemicals, and ENMs (Koumaki et al., 2021). In addition to sewage sludge, contaminated sources of irrigation water can also add PPCPs into the soil system. Veterinary drugs are exposed to the environment through farm wastes, and these are used in agriculture as manures. The residues of the drugs used for livestock treatment and their metabolites thus enter in the food chain. As the environmental impacts of PPCPs are still not completely understood, they are measured as contaminants of emergent concerns (Vasilachi et al., 2021).

The PPCPs have in general low volatility with high polarity and are hydrophilic in nature, and so, they enter into the environment through aqueous transport and later in the food chain (Caliman & Gavrilescu, 2009; Ebele et al., 2017). The physical and chemical properties of the ingredients of the PPCPs also influence their transport into the environment. The biosolids applied to the crop field and the crops irrigated with wastewater can facilitate entry of pollutants into the food chain (Pedersen et al., 2005). Wastewater treatment plants (WWTPs) cannot totally remove PPCPs (Patel et al., 2019). Further, the leachates and runoff of water from landfill can pollute surface and groundwater (Kleywegt et al., 2007). When PPCPs are released in the environment, degradation and transformation processes start. Solar radiation causes photodegradation. Further, based on the depth of water, season, and composition of the materials, the PPCPs are degraded (Alexy et al., 2004). Microbes present in the environment play a role in the biodegradation process as they take part in wastewater treatment processes (Helbling et al., 2010). Microbes use PPCPs' substrates and increase in number for degradation of the PPCPs. In the wastewater treatment process, the PPCPs are partly or fully destroyed or to some extent transformed to metabolites, and the process largely depends on the composition of PPCPs (Xia et al., 2005).

Increasing application of pharmaceuticals in humans and animals and their subsequent entry into soil and aquatic ecosystems makes them emerging contaminants. Pharmaceuticals may have an ecotoxicological effect on non-target species having the same active sites (Lei et al., 2015). Most pharmaceuticals are present in environments at low concentration that do not cause acute toxic effects. However long-term exposure to such low doses in their lifetime results in significant chronic toxic effects (Lei et al., 2015). The veterinary drugs used for animals are also released into the surrounding in bioactive form through excretion which may result in long-term adverse effects on soil, water, animals, plants, etc., and these may also adversely affect human health through food chain contamination.

1.5 Micro- and Nanoplastics

Microplastics and nanoplastics are considered as emerging pollutants which are to be recognized as emerging threats (Liu et al., 2021). Microplastics are plastic substances of 0.1 μm to 5 mm in size, while size of nanoplastics ranges between 0.001 and 0.1 μm (Caputo et al., 2021). Microplastics are found in the environment in the form of fragments, granules, fibres, and spheres or microbeads (Velimirovic et al., 2021), while the shape of nanoplastics in the environment is still unknown because of methodological complications involved in their detection and characterization. Fragments of these micro- as well as nanoplastics are so small that they are neither visible to naked eye nor under a simple optical microscope.

Microplastics and nanoplastics are added to the soil through natural as well as anthropogenic activities. Natural processes include atmospheric inputs (Allen et al., 2019) and flooding with lake and river water (Horton et al., 2017). Various agricultural practices, namely, polyethylene mulching, irrigation with plastic polluted waste water, application of soil amendments containing plastics, and illegal waste dumping are the major anthropogenic activities which introduce micro- and nanoplastics to the soil (Sarkar et al., 2022; Ng et al., 2018).

Microplastics are further broadly grouped as primary and secondary microplastics (Duis & Coors, 2016). Primary microplastics are directly formed from resins and pellets which are utilized in the plastic industry or plastic exfoliating products such as body and face scrubs. On the other hand, secondary microplastics are formed by the fragmentation and weathering process of larger size plastic materials (Lambert & Wagner, 2016).

Various weather parameters such as ultraviolet radiation, alternate wetting and drying periods, and oxidation reduction reactions help in fragmentation of plastic materials. Anthropogenic factors, e.g. mechanical action by machineries during tillage, result in production of microplastics as well as nanoplastics, and their size depends on thickness of layer and heterogeneity with respect to surface of the plastic material. During the fragmentation and disintegration process of formation of microplastics and nanoplastics, there is mechanical alteration. As a result of which, various chemical groups are exposed and bind to exogenous chemicals impacting the plastic degradation rate (Fotopoulou & Karapanagioti, 2012, 2019).

The behaviour of plastics in the soil mainly varies based on their physical properties even though they differ in chemical composition. Crystalline structure in fact is one of the main factors influencing the reactivity of plastics. Plastics having low values of crystalline-to-amorphous ratio are more reactive compared to those having higher values. Higher pore surface and adsorption of chemicals of plastics having low values of crystalline-to-amorphous ratio make them more susceptible to degradation and formation of micro- and nanoplastics. Nanoplastics are chemically more reactive and mobile than microplastics due to higher specific surface area (Hu et al., 2019; Gigault et al., 2018).

Plastics are rapidly colonized by microbes mainly bacteria and form biofilms commonly known as bacterial biofilms (Zettler et al., 2013), and these bacterial biofilms increase chemical adsorption by microplastics (Fotopoulou & Karapanagioti, 2019). Size of nanoplastics is even smaller than most bacteria and viruses, and therefore, no biofilm is seen in the case of nanoplastics. However, pathogens get associated with nanoplastics.

Mostly in the case of microplastics, the major risk associated is they get stuck in the guts of living organisms. However, due to very small size of nanoplastics (i.e. less than 50 nm), they penetrate both prokaryotic and eukaryotic cell membranes with absorbance of pathogenic microbes, thus increasing the biohazard potential of the pathogens. As both microplastics and nanoplastics transport pathogens and chemicals in a passive manner, they are known to show the so-called Trojan horse effect (Zettler et al., 2013; Galloway et al., 2017; McCormick et al., 2014). This Trojan horse effect of micro- and nanoplastics is associated with potential and neglected health hazards for human beings. Microplastics and nanoplastics contaminate the soil and alter physical, chemical, and biological properties of the soil (Pathan et al., 2020; Rillig et al., 2017; Kasirajan & Ngouajio, 2013).

1.6 Agrochemicals

Agrochemicals are the synthetic chemical substances that are being used in agricultural production. Agrochemicals mainly consist of two broad groups, i.e. synthetic fertilizers and pesticides. Most of the fertilizers used in agriculture are inorganic chemicals (except urea), while pesticides are organic in nature. Insecticides, herbicides, fungicides, bactericides, acaricides, etc., are broad categories of pesticides being used to control respective pests. Based on the chemical structure, pesticides are further classified into organo-phosphorous, phenols, phenolics, organochlorines, metallo-organics, nitrogen benzenes, etc. Modernization of agriculture resulted in increased use of both fertilizers and pesticides. However, non-judicious and imbalanced use of these chemical substances has emerged as a potential soil and water contaminant.

Today, on a worldwide basis, there are approximately 1000 pesticides consisting of 800 active ingredients in use, and the number is still increasing (Zhang et al., 2019). Only 1% of applied pesticides is used for pest control that means it reaches the target site of the pest, and the remaining portion accumulates in surrounding soil, water, and air and ultimately enters food chain and affects various non-target species including human beings (Lozowicka et al., 2016; Rani, 2015). Around 80–90% of the applied pesticides to agricultural crop fields contaminate soil, water, air, and non-target plants either directly or indirectly through the process of runoff and drift from on-site pesticide accumulation. Half of this 80–90% of chemical residues are further converted through microbial or chemical processes (known as transformation products) which persist in the ecosystem for more than 10 years.

The adverse impacts of indiscriminate use of these agrochemicals have resulted in reduction in soil fertility, inhibition of growth of beneficial soil microorganisms (Feld et al., 2015), increased concentration of heavy metal, reduction in quality of groundwater, etc. (Ali et al., 2016), and all these ultimately cause reduction in agricultural crop production.

Herbicides above their recommended dose of application affect growth and behaviour of microorganisms having an influence in change in nutrient dynamics in the soil (Das et al., 2015). Various pre-emergence herbicides, e.g. prodiamine, indaziflam, and isoxaben, reduced the accumulation of various macro- (phosphorus, potassium, sulphur, magnesium) and micronutrients (zinc) in affected plants.

The Stockholm Convention has classified some of the pesticides as persistent organic pollutants, i.e. known as POPs (Stockholm Convention, 2021). These POPs are highly persistent in soil and capable of long-distance transport and adversely impact human health and environment. Chlorinated organic compounds such as DDT, dieldrin, aldrin, endrin, chlordane, toxaphene, heptachlor, endosulfan, and hexachlorobenzene are some of the pesticides which are being classified as POPs by the Stockholm Convention. Pesticides belonging to organophosphorus class are the most frequently found pesticides in contaminated soils and have potential risk to humans as well as wildlife (Ballestas et al., 2016). Because of longer persistence of organochlorine pesticides, they are still detected in soil and water despite being banned for use (He, 2017). The most common causes of pesticide poisoning in the world are carbamates and organophosphates (Huang et al., 2015; Lim et al., 2015).

1.7 Interaction Between Emerging Contaminants and Plants

Based on prime concerns related to health and environmental issues, several nations expressed their concerns on emerging contaminants; however, these pollutants when exposed to the environment cause adverse effects not only to health of humans but also that of plants (Santos et al., 2012). Soil contamination is caused by mining, industrial effluents, chemical farm inputs, and unscientific waste disposal by which plants are directly influenced. Further, penetrating impacts of soil and water pollution by emerging contaminants are presumed to reduce crop productivity by altering the physiological and metabolic activities of the plants, and thus, they can cause loss of arable land and habitats of plants (Santos et al., 2012). Crop plants raised under pharmaceutical-contaminated soil uptake the pollutants and can convert them into more toxic substances through drug-metabolizing enzymes. Pharmaceuticals are potentially harmful to plants as they modify enzyme functioning in plants and alter the physiological activities of the plants (Zrncic et al., 2014; Navratilova et al., 2021; Podlipná, 2022). The reactive and hydrophilic compounds as pharmaceutical pollutants make xenobiotic metabolism in plants more complicated where these react with glutathione, glucose, and amino acids and can produce more toxic compounds, and consumption of the contaminated plants by invertebrates or other animals can be fatal (Wagil et al., 2015; Podlipná, 2022). Organic manures derived from animal wastes are one of the important sources of nutrients to plants, and the organic manures may consist of animal excreta and livestock farm wastes. A portion of the drugs used in animal farms remains in the livestock excreta, and the converted organic manure may also contain the same. The anthelmintics drugs such as thiabendazole, albendazole, mebendazole, flubendazole, fenbendazole, and triclabendazole commonly used for deworming of livestock can be exposed to the environment through the manures and liquid form of animal excreta and could have adverse impacts on plants. Fenbendazole and its metabolites are known to alter the protein and gene expression in *Arabidopsis thaliana* inclusive of other physiological and metabolic processes (Syslova et al., 2019). The physiology and metabolism of soybean are also influenced by anthelmintics drugs (Podlipna et al., 2021).

Pesticides used in agriculture to control insect-pests, diseases, and weeds have harmful influences on soil and environment as residues, and they are absorbed by the plants when exposed. To safeguard the human beings from the toxic effect of pesticide residues in the agricultural crops, the World Health Organization (WHO) developed internationally accepted maximum residue limits (WHO, 2022). The common symptoms of pesticide toxicity to plants are necrosis, chlorosis, burns, and deformation of leaves (Sharma et al., 2019). The sensitive plants are more vulnerable to pesticide toxicity. The oxidative stress caused by pesticides toxicity may result in chlorophyll degradation, protein synthesis, and other metabolic activities (Badr et al., 2013; Kilic et al., 2015; Jiang et al., 2016). Moreover, Shahid et al. (2021) mentioned that agrochemical residues acted both positively and negatively on crop plants. A study mentioned that there was no change in nitrogen and protein content in chickpea grain due to pesticides (Khan et al., 2009). The residue of some herbicides such as pendimethalin, atrazine, and alachlor in soil adversely affected germination of crop seeds (Rajashekar & Murthy, 2012; Moore & Kröger, 2010; Tanveer et al., 2009). Further, reduced plant growth was reported due to presence of herbicide residue in the soil (Nadasy et al., 2000). Herbicides are known to disturb nitrogen metabolism and photosynthesis (Nadasy et al., 2000). Fipronil, an insecticide, reduced seed germination in rice; however, diazinon enhanced it (Rajashekar & Murthy, 2012). Dimethoate also negatively impacted on the physiological processes of crops, namely, disturbance and electron transport chain in photosystem II, and reduced assimilate production (Mishra et al., 2008; Shahid et al., 2021). Chibu et al. (2002) reported that application of pesticide chlorpyriphos increased growth and yield attributes and crop productivity in rice and soybean. Similarly, beneficial effects of insecticides application were recorded by Glover-Amengor and Tetteh (2008) in tomato. The fungicide triazole is known to restrict electrolyte leakage and lipid peroxidation in carrot (Gopi et al., 2007). A study revealed that another fungicide named azoxystrobin boosted superoxide dismutase and catalase-peroxidase activities (Wu & Von Tiedemann, 2002).

Micro- and nanoplastics exposed to the environment interact with the plants. Several studies revealed that they reduced chlorophyll content in the plants such as wheat, maize, grass pea, and others (Lian et al., 2020; Wang et al., 2021; Li et al., 2020). Biochemical activities of plants were also influenced by micro- and nanoplastics where peroxidase (POD), superoxide dismutase (SOD), and catalase (CAT) enzyme activities were boosted and along with malondialdehyde (MDA) content (Azeem et al., 2022). In the case of photosynthesis, different types of impacts (neutral, beneficial, and harmful) were noted (Boots et al., 2019; Lian et al., 2021; Zeb et al., 2021) based on the exposed times. The smaller particles of micro- and nanoplastics remained more harmful (Lee et al., 2013).

ENMs are used in agriculture for various purposes such as pest management as new pesticide formulation, identity preservation, sensors to monitor soil conditions, hydrophilic and hydrophobic materials for water conservation, supply of plant nutrients and agrochemical delivery, and genetic engineering purposes (Aslani et al., 2014). Researchers recorded beneficial, harmful, and inconsequential impacts of ENMs to plants (Muller et al., 2005; Bystrzejewska-Piotrowska et al., 2009; Sohaebuddin et al., 2010). However, there is enough evidence which indicates phytotoxicity also (Aslani et al., 2014). Carbon nanotubes (CNTs) were observed to increase seeding growth in rice (Smirnova et al., 2012), but Al_2O_3 NMs reduced root growth of cucumber, maize, carrot, soybean, and cabbages (Kollmeier et al., 2000; Yamamoto et al., 2001; Tian et al., 2007), while ZnO NMs caused toxicity (Ma et al., 2009; Srivastava et al., 2021). The phytotoxicity was related to particle size and surface area (Aslani et al., 2014), and an enhanced surface area determined the number of reactive groups on the surface (Begum et al., 2011). A positive impact of ENMs such as multiwall CNT, Zn, and ZnO was recorded in favouring germination and seedling vigour of Indian mustard and black gram (Ghodake et al., 2010). In conformity, another study revealed that ZnO impacted adversely on seedling germination and plant growth processes (Sharma et al., 2009). Au ENMs also show phytotoxicity (Bai et al., 2010). Actually, the present phytotoxicity profile of ENMs is speculative, and hence, further research is to be intensified into the direction.

1.8 Future Scope of Study on Emerging Contaminant

1. The persistence and bioavailability of different emerging contaminants needs to be studied thoroughly. Their chemical characterization and its relation to persistence need to be evaluated. Persistence of different emerging contaminants under different environments also needs to be evaluated.
2. The risk of different emerging contaminants on human health should be studied. The risk of entry of emerging contaminants into the food chain and the risk of biomagnification need to be evaluated. Food safety regulation in this regard can be made.
3. The existing waste water treatment system needs to be improved to remove the emerging contaminants effectively. This can help in eliminating the entry of emerging contaminants to soil or aquatic systems and thus save the plants from toxicity.
4. Guidelines should be developed to largely avoid or exclude the use of any such ingredients in personal care products that serve as a potential threat to the environment. The ecotoxicity of ingredients should be taken into consideration while developing personal care products.
5. Regular monitoring of emerging contaminants in soil and aquatic systems should be done. Care must be taken to avoid the buildup of these contaminants.
6. Interaction of the emerging contaminants with soil components, i.e. organic matter, mineral components, and soil moisture, should be studied. As persistence and bioavailability are usually affected by these components, hence it will be interesting to know how the emerging contaminants are behaving in relation to interaction with these components.

7. There is a need for thorough study on the mechanisms of micro- and nanoplastics, engineered nanomaterials, and other emerging contaminants by using transcriptomic, proteomic, and metabolomic tools, and the study will help to understand the dynamics of plant responses and resilience.

1.9 Conclusion

Limited availability of productive land and increasing pressure of feeding an ever-growing population make it even more important to take care of the soil resource. Rising population, changing lifestyle, rapid industrialization and urbanization, unsustainable production and value chain management system, etc., are few of the factors that have led to production of huge amounts of waste and contaminants which are accumulated in soil. Many of these contaminants have only been recently studied, and still a lot needs to be researched upon these contaminants to properly understand their environmental impact. Such emerging contaminants when properly characterized and studied for their potential risks can be properly managed. An integrated management strategy that both avoids the production of these contaminants at source and minimizes the impact of the contaminants in soil can help in minimizing the risks arising from these contaminants.

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Chapter 2
Impact of Emerging Metal-Based NPs on Plants and Their Influence on the Phytotoxicity of Other Pollutants

María Dolores Fernández and Concepción García-Gómez

Abstract Metal-based nanoparticles (NPs) are one of the most manufactured nanomaterials and deserve singular attention given their continuous input to the environment, lack of degradation, and accumulation risk. In agricultural soils, the use of organic amendments and wastewater and the application of nanotechnology are important NP inputs. Metal-based NPs have beneficial applications as fertilizers and increase plant resistance to pathogens and environmental abiotic stressors. Ag-, Zn-, Cu-, Ti-, and Ce-based NPs are the most widely used to improve crop production. NPs can also have negative impacts, including phytotoxicity, lower nutrient content in plants, and soil microorganism toxicity. The potential NP interaction with other soil contaminants, including metals and organic compounds, is a major concern because it can modify the bioconcentration or affect the intrinsic toxicity of both substances with the consequent biological impact on plants. Exposure to NP-contaminant mixtures may induce unexpected toxic effects via several different mechanisms that affect the availability, uptake, and metabolic processes involved in the detoxification and degradation of compounds. However, the mechanisms underlying the effects of the NP-contaminant interaction on joint toxicity are poorly understood. This chapter covers some of the most relevant issues concerning the effects of metal-based NPs on plants.

Keywords Metal-based nanoparticles · Nano-agrochemicals · Co-exposure · Phytotoxicity · Bioaccumulation · Chemical mixtures

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M. D. Fernández · C. García-Gómez (✉)
Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria (INIA), CSIC, Madrid, Spain
e-mail: cgarcia@inia.csic.es

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Abbreviations

CAT	Catalase
DDE	Dichlorodiphenyldichloroethylene
GPX	Glutathione peroxidase
OTC	Oxytetracycline
QD	Quantum dot
QNC	Quinclorac
ROS	Reactive oxygen species
SOD	Superoxide dismutase
TC	Tetracycline
ZVI	Zero-valent iron

2.1 Introduction

Nanoparticles (NPs) are materials with the three dimensions below 100 nm and applications in a variety of sectors (e.g., biomedical, chemical, textile, food, agriculture). Their tiny size confers specific characteristics that can intensify their properties. In the natural environment, a small NP size can increase environmental negative impacts compared to the bulk form, but demonstrated beneficial applications as nano-remediators or nano-agrochemicals have been reported (Yang et al., 2019). NPs' morphology is also crucial for showing toxicity in some cases. The coating of NPs' surface and their encapsulation are common practices that can help their stability and change their reactivity and toxicity (Sturikova et al., 2018; Zeng et al., 2019). In the natural environment, NPs are subject to transformation processes like dissolution, aggregation, reduction/oxidation, sulfidation, and adsorption. Aging also drives NPs' properties, including fate and toxicity (Fernández et al., 2021; Romero-Freire et al., 2017; Jośko et al., 2020; García-Gómez et al., 2020).

Overproduction, use, and abuse of NPs have rapidly led to them be released to several environmental compartments, which increases environmental threats to living organisms. The three major sources of terrestrial plant exposure to nanomaterials are air, water, and soil. In soil, the largest amounts accumulate (up to 1.5%, 7%, and 28% of total NPs' production, respectively) (Liu et al., 2020). Atmospheric NPs can be easily deposited on various plant surfaces and infiltrate the plant system via stomatal apertures and across cuticles. The use of wastewater containing aged NPs is another source of NPs to plants. In agricultural soils, the use of amendments (manures, sludges, etc.) and the application of nanotechnology deliberately enable the input of NPs in agricultural environments. In recent decades, the agriculture has faced a wide range of challenges, such as climate change, salinity and drought, soil pollution, and the increasing food demand for a growing population. The use of nanomaterials in modern agriculture helps to gain maximum output from available resources and contributes to mitigate the aforementioned challengers (Rajput et al., 2021). Nonetheless, to promote sustainable progress, it is necessary to assess NPs' toxicity to non-target organisms at the same time as NPs are being investigated and developed.

It is remarkable that NPs at nontoxic concentrations can still be hazardous because of their interaction with other contaminants present in the environment. Previous studies show that NPs can facilitate the intake of metals and organic compounds in plants and other organisms, which can lead to these chemicals' increased toxicity (Deng et al., 2017; Naasz et al., 2018). In some cases, these indirect effects can be more significant than the direct impacts associated with NPs' exposure. Thus acquiring knowledge of the potential effects that result from the interaction of NPs with co-existing organic and inorganic contaminants is critically important for evaluating and regulating the environmental impacts of NPs on plants.

The plant relation with NPs is very complex, and NPs' absorption mechanisms in plants are still poorly understood. Plant systems provide a route for NP uptake, accumulation, and translocation that depends on the physiological properties, functionalization, and the form of exposure of NPs to plants (Agrawal et al., 2022). One of the most important limitations to impact plant uptake of NPs is particle size. Several studies establish 20–50 nm as the size limit for NPs to penetrate and move to plant tissues. In plants, NPs are firstly adsorbed on the root surface, and root exudates and transporter proteins can participate in uptake processes. Tiny NPs can diffuse through epidermal cell wall pores and enter the apoplastic and/or symplastic flow. The apoplast form takes place outside cell membranes through extracellular spaces, cell walls of adjacent cells, and xylem vessels. The symplastic form involves substances and water moving between the cytoplasm of adjacent cells. Larger NPs are first blocked, which results in osmotic pressure and capillary forces that finally help NPs to reach the endodermis by either crossing the cortex cells and diffusing through the apoplastic pathway or merging on symplastic route to penetrate the vascular system (Lv et al., 2019; Deng et al., 2014). The foliar NPs' application implies crossing the cuticle layer, and uptake occurs via two routes: one for polar solutes by polar aqueous pores (hydrophilic pathway) and another for non-polar solutes via diffusion and permeation (lipophilic pathway) (Pérez-de-Luque, 2017; Ali et al., 2021).The cuticle serves as a primary barrier to prevent NPs larger than 5 nm from entering (Molina et al., 2021). This entrance does not prevent root damage because there is evidence for the transport of NPs from the aerial parts to roots (Chichiriccò & Poma, 2015).

Metal-based NPs are one of the most frequently manufactured nanomaterials due to their widespread uses, including environmental applications. Furthermore, given their non-biodegradable nature, significant amounts of these compounds are expected in soil.

This chapter focuses on the impacts that metal-based NPs have on plants. Indirect effects due to NPs are also discussed, such as changes in the plant-soil environment and the influence of co-occurrence with other soil contaminants like organics, metals/metalloids, and nanomaterials.

2.2 Nanoparticles and the Plant's Environment

These new-age materials have the potential to alter biotic and abiotic systems, alterations that are governed mostly by the concentration and physiochemical properties of NPs. Of these, the most dominant are size, shape, and surface charge. Soil properties, mainly pH, organic matter content, cation exchange capacity, texture, moisture content, etc., have the capacity to modify the reactivity, fate, and, ultimately, the toxicity of NPs (Rawat et al., 2018; Gao et al., 2019; García-Gómez & Fernández, 2019). In soil, NPs may undergo several physical-, chemical-, and biological-mediated processes that lower their bioavailable concentration and, hence, their toxicity. In particular, aggregation, retention, adsorption or desorption, dissolution or precipitation, transformation, interaction with other molecules, or incorporation (ingestion-egestion) by organisms are common processes undergone by NPs in natural environments (Amde et al., 2017). Most of these processes depend on soil pH. Under acidic conditions, metallic NPs are transformed into ionic species at high rates, while alkaline environments help the aggregation of NPs. For example, ZnO NPs are differently reactive in acidic (pH 5.4) vs. alkaline (pH 8.3) soils, which results in positive germination and growth responses of nine plants in alkaline soil, but also in negative responses in acidic soil (García-Gómez et al., 2018c). CuO NPs are more toxic to barley at low pH, which is coincident with greater Cu dissolution from NPs (Qiu & Smolders, 2017). Hetero-aggregation induced by the pH of metal-based NPs with soil components enhances their electrostatic/steric stability, but hinders their diffusion and transport in soil (Dimkpa, 2018; Ju-Nam & Lead, 2016). Aggregation also involves a diminished particle surface made available for the release ions, which results in a lower dissolution rate that can attenuate their effects on biological systems.

Regardless of soil physiochemical properties and NP intrinsic characteristics, other factors influence the impact of NPs on plants. Root secretions contain organic molecules of high- and low-molecular weights (polysaccharides, fatty acids, amino acids, metal ions, etc.) that can modify the environment of the rhizosphere, the associated microbiome, and the fate of metal-based NPs (Ahmed et al., 2021). That is, NPs can be deposited on or adhered to the root surface, they can release free metal ions, and they can even be chemically modified as a result of the acids and oxidizing-reducing components of exudates (Gao et al., 2018; Zhang et al., 2017). Low-molecular-weight acid root exudates in rice largely determine the aggregation, sedimentation, and dissolution of CuO NPs (Peng et al., 2019). In cucumber, the binding of Cu NPs to synthetic root exudates significantly reduces both Cu uptake and accumulation (Huang et al., 2017). ZnO NPs applied to soybean plants transform into Zn^{2+} and Zn-citrate due to the lowering soil pH caused by the organic acids secreted by roots. Fe and Cu NPs precipitate as hydroxides (unavailable to plants) owing to exposure to root exudates (Dimkpa et al., 2015; Gao et al., 2018). In turn, the presence of metal-based NPs on the root surface can change the surface chemistry of roots, root secretions, and rhizosphere microbial composition and can, consequently, affect the uptake of nutrients in plants and soil properties. TiO_2 and

Fe_3O_4 NPs rise cysteine and methionine contents and induce alterations in phosphorous speciation in lettuce and wheat root exudates (Zahra et al., 2015; Rafique et al., 2018). Ag NPs apparently induce changes in the root exudates of wheat, cowpea, and mustard (Pallavi et al., 2016) and increased the abundance of diazotrophic bacteria in soil (Shah et al., 2014), while CuO NPs induced plant growth-promoting bacteria in the rhizosphere of red sage (*Salvia miltiorrhiza*) (Wei et al., 2021).

2.3 Positive Effects of Metal-Based NPs on Plants

At appropriate concentrations, metal-based NPs can promote plant growth. They can facilitate nutrient uptake and enhance the efficiency when acting as fertilizers through their slow release (Madzokere et al., 2021; Bindraban et al., 2015) and have the potential to increase plant tolerance to both pathogens and environmental abiotic stressors. Acting efficiently depends on plant species, type and dose of NPs, application method, and growing media (Ananthi et al., 2020). Of the nano-agrochemicals proposed to increase agricultural productivity, metal-/metalloid-based NPs are the commonest ones. Of these, mostly Zn and Cu oxide NPs, followed by Ti and Fe oxide NPs, are used in numerous commercial applications. Hence vast amounts of them will remain as residues (Ruttkay-Nedecky et al., 2017). In crop protection terms, ZnO NPs, Ag NPs, and Cu-based NPs are the most frequently studied ones for their antifungal and antibacterial toxicity (Worrall et al., 2018; Shang et al., 2019; Khan et al., 2019a).

Many studies have evaluated the efficiency of metal-based NPs as fertilizers (Beig et al., 2022; Adisa et al., 2019). Non-nutrient NPs, such as CeO_2, TiO_2, or SiO_2, have positive impacts on plants. By way of example, TiO_2 NPs enhance seed vigor and enzyme activities in maize (Shah et al., 2021) and increase P uptake in soybean (Hussain et al., 2021). SiO_2 NPs positively affect maize seed germination by making larger amounts of nutrients available after altering the pH and conductivity of the growing medium (Suriyaprabha et al., 2012). Despite this, most fertilizer knowledge pays special attention to those that include micronutrients (Zn-, Fe- and Cu-based NPs). Biofortification by means of nanofertilizers with Zn is an effective method for removing zinc deficiency. ZnO NPs act as a micronutrient source, especially in calcareous soils where the available Zn concentration is generally very low, and slow steady zinc release is needed to adapt and match the plant growth stage (Almendros et al., 2022; Du et al., 2019). FeO NPs applied to lettuce at low concentrations increase the germination rate and root length (Delfani et al., 2014), and Fe_2O_3 improves the root growth of peanut plants (Rui et al., 2016). Cu NPs enhance shoot length in lettuce and also coriander germination (Verma & Khanam, 2020), and CuO NPs significantly improve wheat and maize yields (Seleiman et al., 2020). Recent reviews include tables that compile the fertilizer effect of several metal-based NPs by detailing NP concentration, crops, and impacts (Agrawal et al., 2022; Ahmed et al., 2021).

Additionally, new NPs have been proposed to overcome the impact of abiotic stress factors. Abiotic stress is a crucial global issue, and climate conditions and environmental contaminants are the primary causes of crop yield loss worldwide. The effects of metal-based NPs, along with other pollutants, are discussed in a specific section of this chapter. Regarding climate conditions, NPs play a beneficial role in overcoming both salinity and drought stress in plants by inducing the expression of several genes involved in stress response, such as those that enhance their antioxidant defense, trigger the signaling pathway of phytohormones, or alter root hydraulic conductance and water uptake (Zhao et al., 2020; Sarraf et al., 2022). It has been recently stated that several NPs also possess antioxidant "enzyme-like" activities: CeO_2, Fe_3O_4, and Co_3O_4 NPs imitate catalase (CAT); CeO_2, Fe_3O_4, Co_3O_4, MnO_2, CuO, and Au NPs mimic peroxidase; CeO_2 and Pt NPs mimic superoxide dismutase (SOD) activity (Sarraf et al., 2022; Liu et al., 2021). Other authors point out another possible way by which NPs reinforce plants' self-protection against environmental conditions by demonstrating a noticeable rise in the level of some biochemicals like proline or tryptophan. These amino acids play an important role in osmotic adjustment, stomatal regulation, and reactive oxygen species (ROS) scavenging by protecting plants from dehydration (Helaly et al., 2014; Sun et al., 2020; Ramadan et al., 2022). Some examples of NPs that alleviate climatic stress effects on crops are as follows: ZnO NPs improve salt tolerance in tomato (Raghib et al., 2020) and okra (*A. esculentus*) (Alabdallah & Alzahrani, 2020); Ag NPs relieve saline stress in pearl millet (Khan et al., 2020); doped Fe_2O_3 NPs mitigate drought stress in *B. napus* by decreasing the amount of H_2O_2 and the peroxidation of membrane lipids (Palmqvist et al., 2017); a pretreatment of TiO_2, followed by ZnO NPs, improves wheat tolerance to heat stress by enhancing glutathione peroxidase (GPX) and SOD activities, which allows H_2O_2 levels to lower and membrane stabilization to improve (Thakur et al., 2021). A modern review includes a very comprehensive study about the mechanisms involved in the relation between metal-based NPs and abiotic stress in plants (Sarraf et al., 2022).

An emerging research field is the application of metal-based NPs in agriculture to amplify the production of secondary metabolites in plants. Secondary metabolites are small organic molecules, such as alkaloids, terpenoids, coumarins, phenols, etc., which are derivatives of primary metabolism. They are not necessary for both growth and development, but perform special defensive physiological functions like resistance to diseases and insect pests, adaptation to environmental factors, or participation in biochemical processes related to the crop quality and flavor (Rana et al., 2021; Osbourn, 2000). NPs based on Mn, Cu, Zn, Al, Si, Ti, and Ag have been reported to increase the content of these metabolites. For example, 800 mg kg^{-1} of CuO NPs increases p-coumaric acid content in cucumber by 225-fold, while 100 mM of Ag NPs rises the anthocyanin level in *A. thaliana* by 18-fold (Predoi et al., 2020; Zhang et al., 2022). In addition to their protective function for plants, secondary metabolites promoted by NPs have the potential to be used as active ingredients for different purposes in agriculture, medicine, or food sectors (Rana et al., 2021; Predoi et al., 2020).

2.4 Negative Effects of Metal-Based NPs in Plants

The continuous deposition, low biodegradability, and long persistence of metal-based NPs in soils can adversely impact plants and soil organisms, and once these NPs come into contact with plants, they have the potential to alter plant physiology. The evaluation of NP phytotoxicity is a prior key condition for promoting nanotechnology applications and avoiding potential ecological hazards. The negative effects of metal-based NPs on plants are evidenced by the inhibition of the seed germination index (rate and time), alterations to root elongation, root tip morphology, shoot growth, delayed plant development and yield, and lower nutrient uptake, which cause a significant productivity and crop quality losses (Jan et al., 2022). With some exceptions, metal-based NPs are harmful at much higher concentrations than those expected to be found in the environment and those needed for correct plant development (Coman et al., 2019; García-Gómez et al., 2018c). Special attention should be paid to ZnO and CuO NPs because Zn and Cu are essential elements, and the differences between concentrations that act as fertilizers or toxics are small and depend mainly on both soil characteristics and plant species (Obrador et al., 2022; Baskar et al., 2018). Soil pH plays a fundamental role in the phytotoxicity of NPs of metallic origin. In this context, bean and tomato seeds have been grown in two agricultural soils of pH 8.3 and 5.4, in lysimeters, containing ZnO NPs' concentrations ranging from 3 to 225 mg Zn kg^{-1} in a greenhouse experiment for 90 days. After 30 days in acid soil, bean plants died regardless of the Zn level, and tomatoes died at the highest dose. On the contrary in calcareous soil, all the tested concentrations allowed normal crop development (García-Gómez et al., 2017). Tables 2.1 and 2.2 provide some examples of phytotoxicity of non-essential and essential metal-based NPs, respectively. Given the large number of found results, only references published in recent years are cited.

Loss of nutritional value of the edible plant part is a negative issue linked with some metal-based NPs. Exposing tomato plants to several metallic NPs (TiO, Ag, Co, Fe_3O_4, CeO_2, and Ni) leads to a reduction in nutrient elements like Mg, P, and S. Exposure of plants to CeO_2 NPs results in the smaller amount of starch, antioxidants, glutelin, iron, lauric, and valeric acid in rice harvest grains and altered Mo micronutrient and sugar and phenolic contents, along with protein fractionation in fruit of cucumber plants (Ananthi et al., 2020).

The previously mentioned visible signs are macroscopic evidence of other biochemical, physiological, and molecular alterations of plant processes due to the stress caused by NPs being present at a high rate. As mentioned in the above section, low levels of metal NPs can increase plants' protective antioxidant mechanisms to limit ROS generation and, hence, oxidative damage. In contrast at high exposure levels, the reaction of NPs with organelles of cells can lead to excessive ROS generation, and cells are unable to maintain normal physiological redox-regulated functions. Excess ROS damages cellular membrane integrity and induces protein denaturation, deficient enzymatic activity, loss of photosynthetic efficiency, and other genotoxic alterations like damaged DNA structure and chromosomal aberrations (Katarína et al., 2021; Yang et al., 2017; Budhani et al., 2019; Tripathi et al., 2017a).

Table 2.1 Summary of studies on the phytotoxic effects of non-essential metal-based NPs in different exposure media

NP type (size)	Experimental conditions	Plant species	Concentration	Effects	Ref.
Ag NPs					
(10 nm)	Sand, 14 d (7-day-old seedlings)	Wheat	0.5, 1.5, 2.5, 3.5, 5 mg kg^{-1}	Reduced the length of shoots and roots in a dose-dependent manner	Dimkpa et al. (2013)
(20 nm)	Agar (germinated seeds)	Rice	0.3, 60 mg L^{-1}	Reduced the fresh shoot and root weights and root elongation Decreased Chl b and increased carotenoid content	Mirzajani et al. (2013)
(20 nm)	Soil, 120 d (seeds)	Bishop pine	350 and 790 mg kg^{-1}	30 d: reduction of the root length with small effect on aboveground plant biomass at 790 mg kg^{-1} 120 d, all Ag NP levels: reduction of root and shoot biomass and lack of lateral root development	Sweet and Singleton (2015)
(22 nm)	Germination in Ag NPs' solution, 3 d, followed by hydroponics, 14 d	Pea	0.5, 1, 3 mg L^{-1}	0.5 mg L^{-1} did not affect plant parameters. 1 and 3 mg L^{-1} reduced the length and fresh weight of root and reduced the fresh and dry weigh or shoot. Chlorophyll and carotenoids decreased and ROS, MDA contents, and SOD, and APX activities increased	Tripathi et al. (2017b)
(10 nm)	Petri dish, 5 d (seeds)	Wheat	1, 10 mg L^{-1}	Decreased the shoot and root length and fresh biomass 10 mg L^{-1} altered the expression of several proteins mainly involved in primary metabolism and cell defense in plants	Vannini et al. (2014)
(8 nm)	Ag NPs in Hoagland solution, 14 d (seedlings)	Turnip	1, 5.0, 10.0 mg L^{-1}	1 mg L^{-1} did not affect plant parameters. 5 and 10 mg L^{-1} decreased seed germination, growth, biomass, and chlorophyll contents of plant. Induced DNA damage and elevated ROS production	Thiruvengadam et al. (2015)
(10 nm)	Ag NPs' solutions in Petri dish, 14 d (seeds)	Pea	20, 40, 80, 160 mg L^{-1}	Decreased seed germination rate, root fresh weight, and number of secondary roots in a dose-dependent manner. Root dry weight did not change. Shoot length and weight were significantly reduced at the higher doses	Labeeb et al. (2020)

Ce NPs					
(6–48 nm)	Soil (pH 7.8), 84 d (seeds)	Common bean	50, 100, 200 mg kg^{-1}	Reduced fresh weight and increased the mineral contents of the green pods. No effect on the organic nutrient contents Increased stomatal conductance. Decreased antioxidative defense	Ma et al. (2020)
CeO_2 NPs					
(10–30 nm)	Foliar application at 45 d and 52 d post sowing (flowering stage), 57 d in greenhouse	Mung bean	250, 500, 1000 mg L^{-1}	250 and 500 mg L^{-1}: increased dry matter and ROS and stimulated antioxidant enzyme activity 1000 mg L^{-1}: delayed plant growth. Strong accumulation of ROS, enlargement of starch granules and swelling of chloroplasts, reduction of photosynthetic pigments, and chlorosis	Kamali-Andani et al. (2022)
(8 nm)	Soil, 30 d (seeds)	Cilantro	125, 250, 500 mg kg^{-1}	No negative effects were observed in germination and growth. All doses hanged the plant's nutritional status (carbohydrate in the shoots) and increased CAT (shoot) and APX (root)	Morales et al. (2013)
(20–200 nm)	Two soils: RS (residential, pH 5.9) and AS (agricultural, pH 6.7), 28 d (7-day-old seedlings)	Soybean Corn Lettuce Zucchini	500, 1000, 2000 mg kg^{-1}	RS, 500 mg kg^{-1}: increased root biomass of soybean 1000 mg kg^{-1}: reduced shoot and root wet biomass in corn 2000 mg kg^{-1}: reduced shoot wet biomass of lettuce and zucchini and root fresh weight of corn AS: Fresh biomass of lettuce shoots increased at 500 mg kg^{-1}. Root wet weight of zucchini increased at 2000 mg kg^{-1}	Servin et al. (2017a)
(8 nm)	Soil, 48 d in greenhouse (18-day-old seedlings)	Soybean	100, 500, 1000 mg kg^{-1}	500, 1000 mg kg^{-1}: leaves showed visible damaged after 29 days of exposure and increased ROS. Leaf chlorophyll concentrations were not affected	Priester et al. (2017)
(8 nm)	Soil, entire life cycle study in field, 210 d (seeds)	Wheat	100, 400 mg kg^{-1}	No effects on final biomass and yield. No effects on Ce concentration in shoots and sugar and starch contents in grains Grain protein increased (all doses). The chlorophyll content decreased, and CAT and SOD activities increased at 400 mg kg^{-1}	Du et al. (2015)

(continued)

Table 2.1 (continued)

NP type (size)	Experimental conditions	Plant species	Concentration	Effects	Ref.
(25 nm)	Soil, 60 d in greenhouse (seeds)	Ragged-robin	20, 200 mg kg^{-1}	Decreased root and stem biomass. There was no affectation of leaf area and leaf dry weight	Lizzi et al. (2021)
(8 nm)	Soil, 40 d entire life cycle study in greenhouse	*Radish*	62.5, 125, 250, 500 mg kg^{-1}	Germination delayed. Antioxidant capacity increased	Corral-Diaz et al. (2014)
TiO_2 NPs					
5 nm	Soil, life cycle study, 145 d (seeds)	Peanut	50, 500 mg kg^{-1}	Grain weight slightly lowered. Resveratrol content (response to stress) of grains increased Additionally, 500 mg kg^{-1} reduced the total amino acid content	Rui et al. (2018)
24 nm	TiO_2 suspensions in Petri dish, until 2 cm root length in controls (seeds)	Cabbage, carrot, corn, cucumber, lettuce, oats, onion, ryegrass, soybean, tomato	250, 500, 1000 mg L^{-1}	TiO_2 NPs decreased germination in cucumber and soybean but increased in cabbage. The average root length was higher in cucumber and ryegrass but shorter in cabbage, corn, lettuce, and oats related to controls	Andersen et al. (2016)
20–30 nm	Soil, two foliar applications (at 40 and 80 d post sowing), 125 d life cycle study of sowed seeds in field	Sunflower	2.6 mg L^{-1}	No negative effects were observed on the number of plants and seed heads per hectare. However, head diameter, weight of dry seed head, dry seed head weight, and grain yield increased, as well as oil content in the treated plants	Kolenčík et al. (2020)
25 nm	Petri dish, 21 d (seeds)	Tobacco	100, 1000, 25,000, 50,000 mg kg^{-1}	All concentrations reduced root length and plant biomass as TiO_2 NPs increased. At the highest dose, germination rates and the number of leaves decreased	Frazier et al. (2014)

Table 2.2 Summary of studies on the phytotoxic effects of essential metal-based NPs in different exposure media

NP type (size)	Experimental conditions	Plant species	Concentration	Effects	Ref.
Cu NPs					
(15–30 nm)	Soil, life cycle study. Cu NPs were added to soil after seedling emergence	Wheat	25 mg L^{-1}	The spike length decreased, but the number and the weight of grains improved. The SOD activity and sugar content increased	Yasmeen et al. (2017)
(100 nm)	Soil, 60 d (seeds)	Oregano	50, 100, 200 mg kg^{-1}	50 mg kg^{-1}: Increased root and shoot Cu content. Reduced shoot biomass. Not affected shoot length, MDA, or chlorophyll. Increased water content. Modified root and shoot Ca, Fe, Mg, and Mn All Cu NPs' treatments decreased starch, total sugar, and sugar in leaves	Du et al. (2018)
(100–1000 nm)	Soil, 35 d (seeds)	Cilantro	20, 80 mg kg^{-1}	No effect on germination. Shoot length was reduced at 80 mg kg^{-1}. Micro and macro elements in shoot and roots were differently affected: Mn and Ca increased at 20 mg kg^{-1}, but B, Zn, Mg, P, and S reduced at both doses	Zuverza-Mena et al. (2015)
CuO					
(10–100 nm)	Soil, 35 d (seeds)	Cilantro	20, 80 mg CuO kg^{-1}	Both concentrations decreased seed germination. In shoots, both doses reduced Mn, P, and S content but B was increased at 80 mg kg^{-1}	Zuverza-Mena et al. (2015)
(30 nm)	Hydroponics, 10 d (4-day-old seedlings)	Transgenic cotton	10, 200, 1000 mg CuO L^{-1}	200, 1000 mg L^{-1}: reduced shoot and root length, number of root hairs, and Mg, Ca, Mn, Mo, B, and P contents in shoot 1000 mg L^{-1} reduced shoot biomass	Nhan et al. (2016)
(30–50 nm)	8-month aged soil, 80 d (seeds)	Spring barley	300, 2000, 10,000 mg CuO kg^{-1}	2000, 10,000 mg kg^{-1}: decreased root length, plant height, stem, spike length, and grain weight	Burachevskaya et al. (2021)

(continued)

Table 2.2 (continued)

NP type (size)	Experimental conditions	Plant species	Concentration	Effects	Ref.
(25 nm) (50 nm) (250 nm)	Soil (outdoor microcosm), life cycle study, 120 d (seeds)	Soybean	50, 100, 200, 500 mg CuO kg^{-1}	The two highest concentrations reduced seed production regardless NP size. CuO NP size and concentration influenced lipid peroxidation and antioxidant biomarkers	Yusefi-Tanha et al. (2020a)
(40 nm)	Soil, life cycle study, 145 d (seeds)	Peanut	50, 500 mg CuO kg^{-1}	Fresh shoot biomass and grain weight decreased at 500 mg kg^{-1}. Both concentrations reduced total amino acid content and increased resveratrol content (response to stress) of grains	Rui et al. (2018)
Fe_2O_3					
(20 nm)	Soil, life cycle study, 145 d (seeds)	Peanut	50, 500 mg Fe_2O_3 kg^{-1}	Grain weight decreased at 500 mg kg^{-1}. Both concentrations reduced total amino acid content of grains	Rui et al. (2018)
Fe_3O_4					
(12 nm)	Universal soil, 21 d (seeds)	Sunflower	50, 500 mg Fe L^{-1}	Germination decreased at 500 mg Fe L^{-1} and wet weight of seedlings at 50 mg Fe L^{-1}	Kornarzyński et al. (2020)
ZnO NPs					
(20–80 nm)	Soil, 21 d (seeds)	Wheat, radish, vetch	1000 mg Zn kg^{-1}	Shoot lengths was reduced in vetch Shoot weight was reduced in wheat, radish, and vetch	García-Gómez et al. (2015)
(20–80 nm)	Two soils: CS (calcareous soil, pH 8.3), and AS (acidic soil, pH 5.4), 35 d (seeds)	Bean Corn Radish Wheat Cucumber Beet Lettuce Tomato Pea	20, 225, 450, 900 mg Zn kg^{-1}	In CS: ZnO NPs reduced both seed germination of bean at 450 and 900 mg Zn kg^{-1} and shoot growth of wheat, cucumber, and beet at 900 mg Zn kg^{-1}. All doses increased the shoot fresh weight of corn and radish In AS: Decreased seed germination of bean, lettuce, tomato, and beet. Improved germination rate of radish. Declined shoot growth of all plant species at all ZnO NPs' concentrations except pea	García-Gómez et al. (2018c)

(20–80 nm)	Two soils: CS (calcareous soil, pH 8.3), and AS (acidic soil, pH 5.4) in greenhouse, life cycle study, 90 d (seeds)	Bean Tomato	3, 20, 225 mg Zn kg^{-1}	In CS: Increased photosynthetic pigments and soluble protein content In AS: Tomato plants died 1 month after sowing. Chlorophyll and carotenoids decreased in bean. Increased of ROS production at 225 mg Zn kg^{-1} in bean. Soluble protein levels were not affected	García-Gómez et al. (2017)
(10 nm)	Soil, 48 d (18-day-old seedlings)	Soybean	50, 100, 500 mg ZnO kg^{-1}	Leaves resulted damaged after 35 days at 50 and 100 mg Zn kg^{-1}. Leaf total chlorophyll decreased with increasing Zn dose. Genotoxicity was evidenced at 500 mg Zn kg^{-1}	Priester et al. (2017)
(38 nm) (59 nm) (500 nm)	Soil, life cycle study, 120 d, outdoor mesocosm (seeds)	Soybean	40, 80, 160 and 400 mg Zn kg^{-1}	Seed yield increased up to 160 mg kg^{-1} (max for 38 nm size) but decreased at 400 mg Zn kg^{-1} H_2O_2 and MDA content and antioxidant enzyme (SOD, CAT, POX) activities decreased as doses increased up to 160 mg kg^{-1}, but increased at 400 mg Zn kg^{-1} (all sizes except 500 nm, this did no change)	Yusefi-Tanha et al. (2020b)
(<100 nm)	Two soils: CS (calcareous soil, pH 7.8), and AS (acidic soil, pH 4.5), 7 d (seeds)	Wheat	125, 250, 500 mg Zn kg^{-1}	In CS: no effects on root and shoot length In AS: dramatic decrease in root growth, whereas shoot growth was little affected	Watson et al. (2015)
(54 nm)	Soil, 90 d	Tomato	300, 600, 1000 mg ZnO kg^{-1}	In the root, ZnO NP significantly reduced the antioxidant enzyme activities (APX, SOD) and increased H_2O_2 and ROS content. CAT activity was enhanced In leaves, SOD activity increased in all doses and APX at 1000 mg kg^{-1}. CAT activity and total chlorophyll content were reduced In fruits: reduction of phenols, flavonoids, beta carotene, and lycopene	Akanbi-Gada et al. (2019)

(continued)

Table 2.2 (continued)

NP type (size)	Experimental conditions	Plant species	Concentration	Effects	Ref.
Size no detailed	Sand (watering with ZnO NPs' suspensions), 14 d (21-day-old seedlings)	Tomato	200, 400, 800 mg ZnO L^{-1}	400 and 800 mg L^{-1}: significantly inhibited tomato root and shoot dry biomass 800 mg L^{-1}: reduced Chl a and b content but increased carotenoids. The SOD, CAT, and APX activities increased in a concentration-dependent manner	Wang et al. (2018b)
(58 nm)	1-year aged soils: CS (calcareous soil, pH 8.3), and AS (acidic soil, pH 5.4), 90 d, life cycle study (seeds)	Pea Beet root	20, 225 mg Zn kg^{-1}	In CS: no effects on beet. In pea, stem weights were seriously affected at 225 mg Zn kg^{-1} In AS: Beet plants died 1 month after sowing at 225 mg Zn kg^{-1}, and pea plants showed a decreased in stem, leaf, and fruit weights	Obrador et al. (2022)
(58 nm)	1-year aged soils: CS (calcareous soil, pH 8.3), and AS (acidic soil, pH 5.4), life cycle study, 90 d (seeds)	Pea Beet root	20, 225 mg Zn kg^{-1}	In CS: decreased ROS levels and enzymatic activities in leaves. In pea (60 d), soluble protein levels decreased and in beet (90 d) increased In AS: Beet plants died at 225 mg Zn kg^{-1} after 30 d. In pea, the generation of ROS was increased, as well as MDA, GPOD, and APX levels. These effects were not observed on beet	García-Gómez et al. (2018b)

The presence of metal-based NPs in soil may indirectly affect plant growth. The soil environment is a complex system in which each component (soil, soil biota, plants) is interconnected with one another. Some metal-based NPs can increase the abundance of beneficial microbes for soil health and plant development, but even at fertilizer doses, other NPs adversely affect soil microbiota. These are microbes, mainly bacteria and fungi, with key functions, such as plant growth promoters (rhizobacteria), producers of bioactive molecules, or those involved in cellulose/lignin degradation processes (Ameen et al., 2021).

NPs may also impair the soil microbiome involved in biogeochemical processes, mainly the degradation of organic compounds and the recycling of nutrients, including N, P, S, and C, which can ultimately affect plant development (García-Gómez et al., 2018a). Recent reviews include detailed data about the effects of several metal-based NPs on soil and beneficial plant-associated microorganisms (Ameen et al., 2021; Kalwani et al., 2022). For example, Ag NPs affect the symbiotic relation between fava bean (*V. faba*) with *R. leguminosarum* or *G. aggregatum* or a combination of both cultures. Moreover, Ag NPs significantly stunted nitrogenase activity, nodulation, mycorrhizal colonization, and glomalin content (Abd-Alla et al., 2016). Similarly, TiO_2 NPs disrupt the Rhizobium–legume (garden pea) symbiosis system. TiO_2 NPs induce morphological changes in pea roots, such as delayed nodulation development, which hence lead to the onset of nitrogen fixation and damage to the cell surface of *Rhizobium leguminosarum* (Fan et al., 2014).

NPs can indirectly impact the plant growth and development due to the combined action with other contaminants present in the exposure media. This issue is of major concern and is dealt with separately in the next section.

2.5 Nanoparticle Interactions with Co-existing Contaminants

The co-existence of NPs and other contaminants in the environment may result in unexpected toxic effects and changes in the accumulation of both NPs and convectional contaminants in plants. The majority of the works published in the literature deal with the influence of NPs on the toxicity/accumulation of these contaminants. The impact of other contaminants on NPs toxicity is examined to a lesser extent, although these studies are increasing in number. The third group of studies focuses on the joint toxicity of both pollutants (NPs and conventional contaminants) by taking into account the mutual interaction of chemicals in the biological effects of the mixture. Joint toxicity can be similar (additive), stronger (synergistic), or weaker (antagonistic) than that expected from the toxicity of individual components. The application of mathematical models, based on a two-factorial analysis of variance (ANOVA), an isobologram analysis, and toxic unit indices, allows the type of interaction to be determined (Cedergreen, 2014; Uwizeyimana et al., 2017). The application of these models to evaluate the joint toxicity of NP-chemical mixtures to plants is still scarce, although some exceptions exist (Ma et al., 2017).

The combined action of NPs and contaminants depends on several factors, such as the intrinsic properties of NPs and chemicals, crop species, experimental conditions (hydroponic or natural soil media), and exposure mode (direct to soil, foliar, seed treatment, etc.). Most studies have investigated the joint toxicity and bioaccumulation of metal-based NPs and co-contaminants on plants under hydroponic conditions, although the tests conducted with natural soil provide the most reliable data. These tests generally measure traditional endpoints, such as germination, growth, and development, as well as biomarkers of oxidative stress. Gene and protein expression measurements (Pagano et al., 2017), DNA alterations (Zhu et al., 2019), and metabolic profile changes (Lian et al., 2020) have been investigated to a lesser extent, even though they may help to reveal the mechanisms of interaction between contaminants. These assessments with chemical mixtures are generally made at much higher concentrations than realistic environmental concentrations to observe significant toxicity. They are also carried out with pristine nanomaterials despite NPs in the environment being subject to transformation processes (aging), which can affect the interactions of NPs with co-existing contaminants and, hence, their accumulation and toxicity (Jośko et al., 2021a; Servin et al., 2017b). However, studies with environmentally transformed NPs are very scarce.

2.6 Mechanisms Underlying the Influence of NP-Contaminant Interaction on the Joint Toxicity

The mechanisms that underlie changes in toxicity due to co-exposure are complex, scarcely investigated, and poorly understood, especially those conducted in soil. They involve several processes that can individually or simultaneously occur. Combined exposure can alter both availability and degradation in exposure media, modify uptake and internalization in plant cells, and modulate the metabolic processes related to the mechanisms of action, detoxification, and excretion of components from mixtures (Naasz et al., 2018; Deng et al., 2017). Figure 2.1 summarizes the relevant mechanisms.

Many studies attribute co-exposure effects on the toxicity and accumulation of NPs and contaminants to changes in the availability of chemicals for organisms (Khan et al., 2019b; Zhang & Zhang, 2020; Adrees et al., 2020). NPs are characterized by high reactivity, a large specific surface area, and strong adsorption capacity. If contaminants are adsorbed to NPs or held in precipitating NP aggregates, the availability and bioaccumulation of these co-existing contaminants are likely to reduce (Bao et al., 2019; Ma et al., 2017). Co-contaminants can modify surface properties and/or transform the functional groups that coat NPs, which lead to changes in their electronegativity and promote the formation of homo- and hetero-aggregates of NPs (Xiao et al., 2021). Both aggregation and adsorption processes can reduce bioavailability and slow down the dissolution of metal-based NPs and, therefore, the release of metal ions (Xiao et al., 2022) with consequent effects on plant toxicity. Indirectly co-present heavy metals can also induce excretion of root exudates, which affects NP aggregation (Sharifan et al., 2020). In soil, complex interactions (adsorption, competition) occur among NPs, co-contaminants, soil

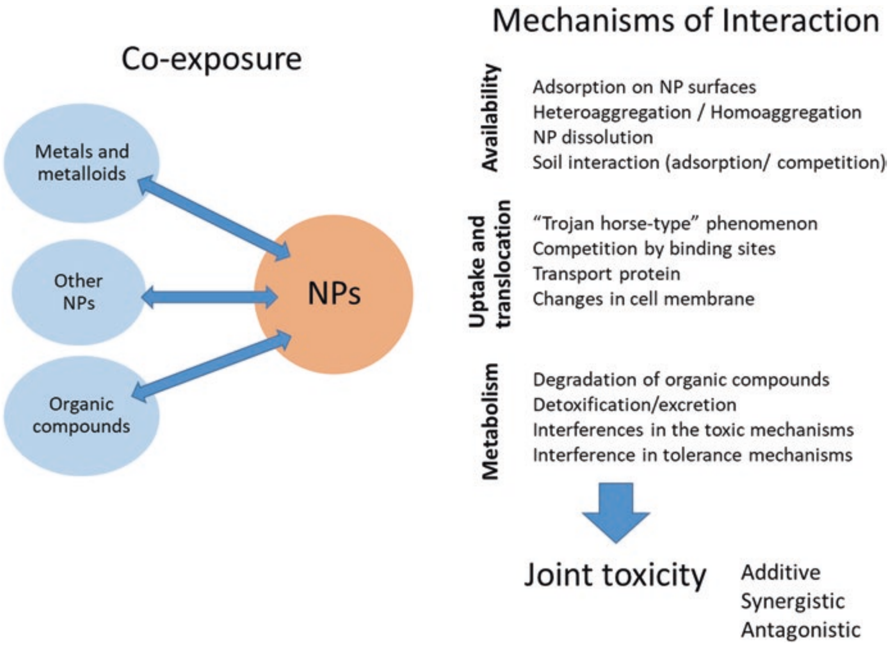

Fig. 2.1 Potential mechanisms responsible for the interaction between NPs and other co-contaminants in plants

particles, and organic matter. The NPs and metals released from them can compete with other metals and contaminants for sorption sites, which might alter the availability of NPs and chemicals for plants under co-contamination conditions (Zhang et al., 2019; Naasz et al., 2018). NPs can also affect the formation of soil aggregates and can, thus, indirectly change heavy metal distribution in soils and their availability (Zhang & Zhang, 2020). In turn, NPs can be modified by edaphic and soil biotic factors.

The second mechanism of interaction focuses on the processes related to the uptake and translocation of metals (Skiba et al., 2020; Sharifan et al., 2020) and organic contaminants (Bao et al., 2019; De La Torre-Roche et al., 2013) in plants as a result of co-exposure. The uptake of xenobiotics by plants can be affected because the contaminants in the mixture can (i) compete for the same transporters and binding sites on the cell membrane; (ii) modify hydrophobicity or damage the cell membrane's physical integrity; and (iii) alter the performance of membrane transport proteins, as well as the metabolic processes involved in the uptake and sequestration of substances in cellular compartments. In addition, the adhesion of NPs to the root surface can act as a physical barrier, which can hinder the uptake of other substances by plants. Organic compounds can change the electronegativity or affect the coating groups on the NP surface, which affects the nano-interaction with organism/cell surfaces.

Adsorption of metals and organic compounds on NP surfaces can display dual behavior with contradictory consequences. NPs can act as carriers of chemicals, which facilitate the entry of substances in cells (the Trojan horse-type phenomenon) (Naasz et al., 2018). Once inside organisms, the subsequent release of adsorbed contaminants can enhance the phytotoxic effects of these substances. Conversely, sorption of compounds to NPs can prevent chemicals from accumulating in plants if NPs reduce the availability of contaminants, as indicated above, or the NP-compound complex is negligibly internalized by plants. A third scenario can occur, in which NPs facilitate the uptake of compounds, but sorption is irreversible, and compounds remain attached to NPs inside organisms. In these cases, the toxicity of NP-contaminant combinations can be expected to diminish.

Finally, the impact of combined exposure can result from alterations to the metabolic processes involved in toxicity and detoxification mechanisms or those related to tolerance to contaminant stress (e.g., antioxidant enzymes involved in oxidative stress tolerance) (Jośko et al., 2021a; Kamali-Andani et al., 2022; Rizwan et al., 2019a, b). NPs can increase the toxicity of organic compounds by either facilitating transformation to compounds being more toxic than parents or hindering the interior degradation rate of these organic compounds and their excretion, which can imply a higher compound concentration in organisms (Deng et al., 2017).

2.7 Effects of Combined Exposure to NPs and Co-existing Contaminants on Their Accumulation and Toxicity to Plants

In plants, the interaction of NPs with pre-existing contaminants leads to changes in their biological effects (bioaccumulation and/or toxicity). Most studies have observed reduced chemical accumulation in plants in the presence of NPs (Hussain et al., 2019; Rizwan et al., 2019a, b). However in some exceptions, NPs promote the accumulation of metal ions (Xiao et al., 2022; Venkatachalam et al., 2017) and organic compounds (Bao et al., 2019). Changes observed in the bioaccumulation of contaminants do not always correlate with changes in toxicity in plants. Negative biological effects generally decrease with declining bioaccumulation (Ma et al., 2017; Ji et al., 2017; Hussain et al., 2019). In some cases, enhanced chemical accumulation in the presence of NPs does not lead to greater toxicity compared to individual treatments (Venkatachalam et al., 2017; Zhang et al., 2019). Some studies report changes in toxicity upon mixture exposure, but no changes in accumulation (Haisel et al., 2019).

2.7.1 The Interaction Between NPs and Metal/Metalloid

The phytotoxic effects of co-exposure to NP-contaminant mixtures on plants are tested mainly with metals as co-contaminants where the combination with Cd predominates (Table 2.3). Cd is one of the major pollutants in soils, and it is

Table 2.3 Summary of studies on the effects of NP interaction with metal/metalloid on their accumulation and toxicity to plants in different exposure media

NP type	Co-contaminant	Experimental conditions	Plant species	Concentration	Joint effect	Ref.
Ag	Sb(III)/(V)	Aqueous suspension, 6 d (16-day-old seedling)	Soybean	1 mg Ag L^{-1} 100 mg Sb(III) L^{-1} 100 mg Sb(V) L^{-1}	Synergistic toxicity (growth and stress oxidative) Ag NPs increased and decreased Sb(III) in roots and leaves, respectively, and increased Sb(V) (stem and leaves) Sb(V) increased Ag in plant	Cao et al. (2020)
CeO_2 ZnO	As(III) As(V)	Aqueous suspension, 6 d (42-day-old seedling)	Rice	100 mg Ce L^{-1} 100 mg Zn L^{-1} 1 mg As(III) L^{-1} 1 mg As(V) L^{-1}	ZnO NPs decreased As(III) in roots and shoots and As(V) in roots CeO_2 NPs did not affect As in roots and shoots As(III) and As(V) increased Ce in shoots As did not affect plant uptake of Zn	Wang et al. (2018c)
CuO	Fe(II)	Aqueous solution, 3 d (30-day-old seedling)	Rice	100 mg Cu L^{-1} 3 mM Fe	Fe reduced Cu in roots and shoots	Yuan et al. (2021)
Fe_2O_3	Cd	Soil and foliar exposure in plants grown in Cd-contaminated soil, 125 d (seed sowing)	Wheat	Soil exposure: 5, 10, 15, 20 mg Fe kg^{-1} Foliar exposure: 5, 10, 15, 20 mg Fe L^{-1} 7.38 mg Cd kg^{-1}	Fe_2O_3 NPs enhanced the plant growth, photosynthesis, and grain yield and decreased Cd (grains, shoots, and roots)	Hussain et al. (2019)
Fe_2O_3	Cd	Cd-contaminated soil, under normal and water-limited conditions, 120 d (seed sowing)	Wheat	25, 50, 100 mg Fe kg^{-1} 7.67 mg Cd kg^{-1}	Fe NPs increased the plant growth and photosynthesis and reduced oxidative stress and decreased Cd in wheat grains	Adrees et al. (2020)

(continued)

Table 2.3 (continued)

NP type	Co-contaminant	Experimental conditions	Plant species	Concentration	Joint effect	Ref.
Fe_2O_3 ZnO	Cd	Cd-contaminated soil, 124 d (sowing of 24-hour seed primed)	Wheat	5, 10, 15, 20 mg Fe L^{-1} 25, 50, 75, 100 mg Zn L^{-1} 7.38 mg Cd kg^{-1}	Fe_2O_3 and ZnO NPs increased the plant growth, grain yield, and photosynthesis in Cd stressed plants and decreased the Cd accumulation	Rizwan et al. (2019a)
Si	As(V)	Irrigation of plants grown in peat moss and perlite, 150 d (seedling)	Tomato	250, 1000 mg Si L^{-1} 0.2, 0.4. 0.8, 1.6, 3.2 mg As L^{-1}	Si NPs decreased tomato yield at the highest doses of As and Si Si NPs decreased As translocation at leaves, at all tested concentrations	Gonzalez-Moscoso et al. (2022)
Si	Cd	Soil and foliar exposure in plants grown in Cd-contaminated soil, 124 d (seed sowing)	Wheat	Soil exposure: 300, 600, 900, 1200 mg Si kg^{-1} Foliar exposure: 300, 600, 900, 1200 mg Si L^{-1} 7.38 mg Cd kg^{-1}	Si NPs (soil and foliar exposure) increased the plant growth and photosynthetic pigments and reduced oxidative stress in Cd stressed plants and reduced Cd in shoots, roots, and grains	Ali et al. (2019)
TiO_2	Cd	Kimura solution, 10 d (10-day-old seedling)	Rice	10, 100, 1000 mg Ti L^{-1} 10, 20 mg Cd L^{-1}	TiO_2 NPs decreased Cd toxicity (net photosynthetic rate and chlorophyll content) and Cd in roots and leaves Co-exposure decreased Ti in roots	Ji et al. (2017)
TiO_2	Cd	Root and foliar exposure of plants grown in Hoagland suspension, 14 d (14-day-old seedling)	Corn	100, 250 Ti mg L^{-1} 50 Cd mM	Root co-exposure caused synergistic effects (growth and chlorophyll content) and increased Ti in root and shoot TiO_2 NPs increased and reduced Cd uptake at 100 and 250 mg Ti L^{-1}, respectively Foliar application of TiO_2 (100 mg Ti L^{-1}) decreased Cd toxicity and the Cd accumulation in shoots	Lian et al. (2020)

TiO_2	Cu	Hoagland suspension, 6 d (5-day-old seedling)	Soybean	10 mg Ti L^{-1} 1, 2, 5, 20 mg Cu L^{-1}	TiO_2 NPs enhanced Cu toxicity (growth of root and shoot) and accumulation and inhibited the translocation of Cu, at 1 and 2 mg Cu L^{-1} TiO_2 NPs did not affect Cu toxicity and accumulation at 5 and 20 mg L^{-1}	Xiao et al. (2021)
ZnO	Cu Fe Mn	Hoagland suspension, 12 d (7-day-old seedling)	Pea	100 mg Zn L^{-1} 0.05 mg Cu L^{-1} 0.4 mg Mn L^{-1} 10.28 mg Fe L^{-1}	ZnO NPs increased Cu, decreased Mn, and changed Fe concentration (root and shoot)	Skiba et al. (2020)
ZnO	Cd Pb	Hoagland suspension, 15 d (seedlings)	White leadtree	25 mg Zn L^{-1} 50 mg Cd L^{-1} 100 mg Pb L^{-1}	ZnO NPs decreased Cd toxicity (growth and oxidative stress) and increased the Cd and Pb accumulation	Venkatachalam et al. (2017)
ZnO	Cd	Liquid RH medium, 11 d (35-day-old seedling)	True fox sedge	10, 50 μM Zn 10 μM Cd	ZnO NPs increased Cd toxicity (photosynthetic pigments) and did not affect the Cd accumulation	Haisel et al. (2019)
ZnO	Cd	Soil and foliar exposure in plants grown in Cd-contaminated soil, 125 d (seed sowing)	Wheat	Soil exposure: 25, 50, 75, 100 mg Zn kg^{-1} Foliar exposure: 25, 50, 75, 100 mg Zn L^{-1} 7.38 mg Cd kg^{-1}	ZnO NPs enhanced the plant growth, photosynthesis, and grain yield and decreased the Cd accumulation	Hussain et al. (2018)
ZnO	Cd	Foliar exposure of plants grown in Cd-contaminated soil, 75 d (seed sowing)	Corn	50, 75, 100 mg Zn L^{-1} 7.86 mg Cd kg^{-1}	ZnO NPs improved the plant growth, chlorophyll content, and gas exchange and reduced Cd in roots and shoots	Rizwan et al. (2019b)

(continued)

Table 2.3 (continued)

NP type	Co-contaminant	Experimental conditions	Plant species	Concentration	Joint effect	Ref.
ZnO	Cd	Cd-contaminated soil, under normal and water-limited conditions, 125 d (seed sowing)	Wheat	25, 50, 100 mg Zn kg^{-1} 7.67 mg Cd kg^{-1}	ZnO NPs increased the plant growth, grain yield, and chlorophyll levels, decreased oxidative stress, and reduced Cd in shoots and roots	Khan et al. (2019a, b)
ZnO	Cd	Soil, 60 d (30-day-old seedling)	American pokeweed	500 mg Zn kg^{-1} 10, 100 mg Cd kg^{-1}	Synergistic inhibition of the plant growth and increase of stress oxidative ZnO NPs increased the Cd accumulation	Xiao et al. (2022)
ZnO	Cd	Soil, 120 d (8-cm seedling)	Rice	50, 100, 500 mg Zn kg^{-1} 1.0, 2.5, 5.0 mg Cd kg^{-1}	ZnO NPs (500 mg kg^{-1}) decreased Cd toxicity (shoot and total biomass) and increased Cd in root, shoot, and grain	Zhang et al. (2019)
ZnO	Cd	Soil, 63 d (48-hour germinated seeds)	Sweet sorghum	50, 250, 500 mg Zn kg^{-1} 5 mg Cd kg^{-1}	Co-exposure (ZnO NPs 250 and 500 mg kg^{-1}) showed synergistic effects on the plant growth Co-exposure (ZnO NPs 50 mg kg^{-1}) showed antagonistic effects on the plant growth ZnO NPs significantly decreased Cd in shoot and root The effect of Cd in the Zn accumulation depended on the Zn rate	Wang et al. (2018a)

well-known that it affects the biochemical and physiological plant functions and can accumulate in edible tissues. Many authors have reported data collected from experiments performed in hydroponic media with conflicting results. For example, TiO_2 NPs alleviate Cd toxicity (net photosynthetic rate and chlorophyll content) and decrease Cd uptake in roots and leaves of rice (*Oryza sativa* L.) (Ji et al., 2017). In turn, the presence of Cd significantly decreases Ti accumulation in rice roots. Similarly, Venkatachalam et al. (2017) report that phycomolecule-coated ZnO NPs (25 mg L^{-1}) enhance seedling growth, reverse the oxidative stress symptoms induced by Cd and Pb, and induce desirable genomic alterations in *Leucaena leucocephala*. However, unlike the previous paper, NPs increase Cd and Pb accumulation in plant tissues. An opposite trend is indicated in a hydroponic study with *Carex vulpina* (Haisel et al., 2019), where ZnO NPs at low concentrations (10 or 50 μM of Zn) significantly aggravate the negative effect of Cd, which is reflected mostly in changes in the content of photosynthetic pigments. Exposure mode and contaminant levels are key factors in plant response to co-exposure to NPs and metals according to Lian et al. (2020), who studied the combined effect of TiO_2 NPs and Cd on metal accumulation and toxicity to hydroponic maize (*Zea mays* L.). Root applications of TiO_2 NPs and Cd synergistically inhibit plant growth and development, while the foliar spray of TiO_2 NPs can partially protect plants from Cd stress. Similarly at low Cu concentrations (1 and 2 mg Cu L^{-1}), TiO_2 NPs enhance the toxicity and accumulation of Cu in soybean, whereas the effects caused by the co-presence of TiO_2 NPs disappear at 5 and 20 mg Cu L^{-1} (Xiao et al., 2021). Cu adsorption in TiO_2 NPs increases with a rising Cu concentration, with the subsequent reduction in the zeta-potential, aggregation, and sedimentation of TiO_2 NPs. This fact can lead to a lower Cu and Ti concentration in hydroponic media, and, consequently, Cu toxicity can be alleviated.

The contaminant type also plays an important role in joint bioaccumulation. A hydroponic study with five forms of ZnO NPs (100 mg L^{-1}) reports that NPs alter Cu, Mn, and Fe uptake and translocation in pea (*Pisum sativum* L.) plants, but effects are element-specific (Skiba et al., 2020). Similar behavior has been observed with two non-essential metals (Pb and Cd), where the influence of ZnO NPs on the accumulation of these metals in the edible tissue of three leafy green species is impacted by the co-contaminant nature (Sharifan et al., 2020). Additionally, metal-based NPs with oxidizing or reducing properties can regulate the oxidation states of some metals and, hence, their uptake and toxicity to plants (Cao et al., 2020). Combined exposure to CeO_2 NPs or ZnO NPs and inorganic As species differently affects As(III)/As(V) accumulation and speciation in rice (*Oryza sativa* L.) (Wang et al., 2018c).

The joint toxicity and bioaccumulation of NPs and metals have been also studied in plants growing in soil. A fair number of studies conducted with Cd-contaminated soils indicate that ZnO NPs and Fe_3O_4 NPs applied by different routes (soil exposure, foliar spray, seed priming) mitigate Cd phytotoxicity to wheat (*Triticum aestivum*) (Hussain et al., 2018, 2019; Rizwan et al., 2019a, b). Decreased toxicity has been generally associated with reduced Cd accumulation in plants, which might be due to a drop in available Cd in soil. These outcomes are similar to those obtained in two experiments performed under water-limited conditions (Khan et al., 2019b; Adrees et al., 2020). Si NPs applied directly to soil or as foliar spray also promote yield and reduce Cd accumulation in wheat (Ali et al., 2019). Si NPs reduce Cd accumulation in plants by lowering Cd available concentrations in soil. With foliar applications, diminished Cd accumulation may be due to other causes, such as dilution effects because of increased growth or compartmentation into vacuoles, which restrict metal translocation to grain.

In contrast, co-exposure to ZnO NPs and Cd amplifies toxicity (root cell damage and increased oxidative stress) to *Phytolacca americana* L. (Xiao et al., 2022). In this study, Cd^{2+} promotes the release of Zn ions from ZnO NPs due to the interaction of Cd on NP surfaces, which can explain the increased toxicity of the mixture. In addition, ZnO NPs considerably increase Cd accumulation.

Interestingly, both synergistic and antagonistic effects of the ZnO NPs and Cd mixture appear in sweet sorghum (*Sorghum bicolor*) grown in soil depending on the contaminant concentration (Wang et al., 2018a). The mixture shows synergism at the two highest doses (250 and 500 mg Zn kg^{-1}) of ZnO NPs. ZnO NPs are non-phytotoxic at the lowest dose (50 mg Zn kg^{-1}) and show antagonistic interactions with Cd in plant growth. All the ZnO NPs' doses significantly lower the Cd concentrations in sorghum shoots and roots, whereas the effect of Cd on Zn accumulation depends on the Zn rate. In addition to the application rate, the plant growth stage is an important factor for the biological effects that result from co-exposure. In a soil-rice system (Zhang et al., 2019), the main impact of NPs on Cd toxicity and bioaccumulation appears in the tillering stage, where ZnO NPs ameliorate toxic Cd effects (plant height). However, this effect diminishes over time and disappears in the fruiting stage.

2.7.2 The Interaction Between Different NPs

Only a few studies have investigated the impact of NP mixtures on plants, even though a variety of NPs may co-exist in the natural environment (Table 2.4). Two experiments in soilless culture media have assessed the effects of binary mixtures of metal-based NPs on plants with different results. In a germination assay with five different NPs (ZnO, CuO, TiO_2, Cr_2O_3, and Fe_2O_3) and four plant species (cress, flax, wheat, and cucumber), Jośko et al. (2017) have found that co-exposure at 100 mg L^{-1} exerts significantly less toxicity (root growth inhibition) compared to single exposure and regardless of its components. In another study, binary combinations of five NPs have shown increased or decreased metal content and toxicity to zucchini (*Cucurbita pepo* L.) grown in vermiculite for 21 days depending on NP combinations (Pagano et al., 2017). Both experiments suggest that the differences in toxicity observed between simple and combined treatments, and between different binary NP mixtures, can be explained by distinct solubility and the ratio of the particulate/ionic forms that derive from NPs, as well as greater particle aggregation under combined stress conditions.

Table 2.4 Summary of studies on the effects of the interaction between different NPs' types on their accumulation and toxicity to plants in different exposure media

NP type	Co-contaminant	Experimental conditions	Plant species	Concentration	Joint effect	Ref.
CeO_2 CuO La_2O_3 ZnO CdS QDs	Binary mix	Aqueous solution, vermiculite, 21 d (7-day-old seedling)	Zucchini	NPs 500 mg L^{-1} CdS QDs 100 mg L^{-1}	Synergistic and antagonistic effects on toxicity (growth, photosynthetic pigments, and gene expression) and accumulation depending on binary NP combinations	Pagano et al. (2017)
CeO_2 Se	Binary mix	Foliar application in plants grown in soil, 59 d (seed sowing)	Mung bean	250, 500, 1000 mg Ce L^{-1} 25, 50, 75 mg Se L^{-1}	Se NPs (25 and 50 mg L^{-1}) ameliorated the stress (dry matter, photosynthetic pigments, and antioxidants) of CeO_2 NPs	Kamali-Andani et al. (2022)
CuO Cr_2O_3 Fe_2O_3 TiO_2 ZnO	Binary mix	Aqueous solutions, 3 d (seeds)	Cress Flax Wheat Cucumber	10, 100, 1000 mg L^{-1}	Antagonistic toxicity (root growth) at 100 mg L^{-1} No differences at 10 and 1000 mg L^{-1}	Jośko et al. (2017)
CuO ZnO	Binary mix	LUFA soil, 7 and 30 d (4-day-old seedling)	Barley	300 mg Cu kg^{-1} 300 mg Zn kg^{-1}	Co-exposure caused synergistic and antagonistic effects on toxicity (antioxidative enzyme activity and relative gene expression) depending on plant part and exposure time and increased Cu and Zn in leaves at 30 d	Jośko et al. (2021a)
CuO ZnO	Binary mix	LUFA soil, 7 and 30 d (4-day-old seedling)	Barley	300 mg Cu kg^{-1} 300 mg Zn kg^{-1}	Co-exposure caused changes in mineral composition in leaves and downregulated genes related with metal influx to cells	Jośko et al. (2021b)

Two soil experiments have confirmed the influence of dose and exposure time on the toxicity magnitude of NP mixtures. Kamali-Andani et al. (2022) have observed that Se NPs modify the stress caused by CeO_2 NPs on mung bean (*Vigna radiata*) plants grown under greenhouse conditions, but this effect depends on the foliar application rates of both NPs. The low concentrations of Se NPs (25 and 50 mg Se L^{-1}) improve photosynthesis by increasing antioxidant activity and proline content, which lowers the levels of ROS and lipid peroxidation caused by CeO_2 NPs. Other noteworthy studies indicate that the effects of co-exposure to CuO and ZnO NPs on toxicity and metal accumulation on soil-grown barley (*Hordeum vulgare* L.) vary with exposure time (7 and 30 days), although a general tendency is not easy to identify (Jośko et al., 2021a, b). Their findings reveal that co-exposure results in the downregulation of the genes related to the metal influx to cells. Interestingly, the binary mixtures of CuO and ZnO NPs have antagonistic effects on Zn and Cu availability in soil, whereas mixtures of their metal salts show synergism. Soil-extractable Zn and Cu concentrations weakly correlate with Cu and Zn contents in barley.

2.7.3 The Interaction Between NPs and Organic Compounds

Both decreases and increases in toxicity and contaminant accumulation in plants due to interactions between metallic NPs and organic compounds have been reported (Table 2.5). For example, in an interesting study, Ma et al. (2017) have investigated the joint effects of TiO_2 NPs and tetracycline (TC) on rice (*Oryza sativa* L.) grown in hydroponic media for 10 days. Three mathematical models are applied to toxicity (plant growth, changes in oxidative stress enzymes, and macro-/micronutrient contents) data to establish the type of toxic interaction, i.e., synergistic, additive, or antagonistic, to result from co-exposure. The analyses indicate that TiO_2 NPs and TC antagonistically interact, showing overall phytotoxicity alleviation compared to that expected of the toxicity of individual treatments. Decreased phytotoxicity is accompanied by low TC levels in plants. This is probably due to the sorption of the antibiotic into TiO_2 NPs, which can decrease its availability for rice seedlings. However, Ti levels in rice shoots and roots rise in the combined treatment, which is attributed to the alteration of surface charges of TiO_2 NPs caused by TC. In contrast, hydroponically exposed wheat (*Triticum aestivum* L.) to phenanthrene and ZnO (NPs and bulk) mixtures shows greater toxicity compared to individual treatments (Zhu et al., 2019). This effect is more evident in DNA damage in wheat root cells, especially for ZnO NPs. In another study, the plant response to the combined exposure to NPs and an organic contaminant strongly depends on the concentration of both xenobiotics (Zhang et al., 2020). At low concentrations (50 and 250 mg L^{-1}), zero-valent iron (ZVI) NPs alleviate the toxicity (root length) of quinclorac herbicide (QNC) to *Oryza sativa* L. However, this effect disappears at the high ZVI NPs' concentration (750 mg L^{-1}), which is possibly due to the toxicity of ZVI NPs itself at this concentration. QNC content in both shoots and roots lowers compared to the tissues exposed to QNC alone, probably because ZVI NPs remove QNC from culture solution.

Table 2.5 Summary of studies on the effects of NP interaction with organic compounds on their accumulation and toxicity to plants in different exposure media

NP type	Co-contaminant	Experimental conditions	Plant species	Concentration	Joint effect	Ref.
Ag	Dichlorodiphenyldichloroethylene (DDE)	Hoagland solution, 19 d (5–7-day-old seedling)	Soybean Zucchini	500, 2000 mg Ag L^{-1} 100 µg DDE L^{-1}	Ag NPs decreased DDE in soybean Ag NP (500 mg L^{-1}) decreased DDE in zucchini	De La Torre-Roche et al. (2013)
Cu	Kinetin	Soil, 55 d (seeds)	Bean	50, 100 mg Cu kg^{-1} 10, 100 µM kinetin	Kinetin decreased Cu in roots and increased Cu in leaves	Apodaca et al. (2017)
nZVI	Quinclorac (QNC)	Aqueous solution, 7 d (14-day-old seedling)	Rice	50, 250, 750 mg Fe L^{-1} 5, 10 mg QNC L^{-1}	nZVI (50 and 250 mg Fe L^{-1}) alleviated toxicity (root length) of QNC and reduced QNC in shoots and roots	Zhang et al. (2020)
γ Fe_2O_3	Oxytetracycline (OTC)	Rice nutrient solution, 10 d (16-day-old seedling)	Rice	25 mg Fe L^{-1} 25, 100 mg OTC L^{-1}	Fe_2O_3 NPs increased OTC in root and on root surface and reduced OTC in shoot at 25 mg OTC L^{-1} OTC increased Fe on root surface and in shoot and did not affect Fe in root	Bao et al. (2019)
TiO_2	Tetracycline (TC)	Hoagland solution, 10 d (18-day-old seedling)	Rice	500, 1000, 2000 mg Ti L^{-1} 5, 10, 20 mg TC L^{-1}	Antagonistic effects on phytotoxicity (plant biomass, oxidative stress, and macro-/micronutrient contents) Co-exposure diminished the TC accumulation and increased the Ti accumulation	Ma et al. (2017)
ZnO	Phenanthrene	Hoagland solution, 15 d (6-day-old seedling)	Wheat	250, 500, 1000 mg Zn L^{-1} 1 mg phenanthrene L^{-1}	Co-exposure increased toxicity (plant growth) Phenanthrene increased the DNA damage of wheat root cells caused by ZnO NPs ZnO NPs reduced the phenanthrene accumulation in roots and leaves	Zhu et al. (2019)

Experiments conducted simultaneously with other size particles and metal-based salt are particularly interesting because they allow the role of NP-specific properties in the interaction to be evaluated. Several works have evidenced that NPs' co-exposure with metals or organic compounds can elicit different biological responses in plants to those caused by the other chemical forms. For example, De La Torre-Roche et al. (2013) have demonstrated that the effects caused by Ag NPs on the accumulation and translocation of dichlorodiphenyldichloroethylene (DDE) in soybean (*Glycine max* L.) and zucchini (*Cucurbita pepo* L.) grown in vermiculite differ from those caused by bulk or ionic Ag.

Similarly, the influence of oxytetracycline (OTC) on Fe accumulation in rice tissue (*Oryza sativa* L.) differs for plants exposed to ionic Fe or Fe_2O_3 NPs (Bao et al., 2019). OTC promotes Fe accumulation on root surfaces and shoots in Fe_2O_3 NPs treatments, which is the exact opposite result of Fe-EDTA treatments. The presence of ZnO (NPs and bulk) reduces phenanthrene accumulation in wheat (roots and leaves), but this effect is stronger for NPs than for bulk counterparts. This is probably due a stronger sorption capacity of NPs than bulk material (Zhu et al., 2019). Interestingly, ZnO (NPs and bulk) increases the detrimental effects of Cd on hydroponic *Carex vulpina* L. plants, whereas Zn salt protects plants against Cd-induced toxicity (Haisel et al., 2019). Although these results are not conclusive, they indicate some possible underlying mechanisms related to the NP properties inherent to their size that affect the interaction of NPs with conventional co-contaminants. This fact emphasizes the need to consider the combined action of NPs with other contaminants present in media to assess and regulate the environmental impacts of NP applications.

2.8 Conclusions

Metal-based NPs have many positive effects on plants which encourage their use to improve crop production and sustainable agriculture, although they also have detrimental effects. Among others, they may produce physicochemical soil alterations, modify the rhizosphere environment, and have toxic effects on plants and soil biota, particularly on beneficial microbial populations. Notwithstanding, the demonstrated fertilizing effects of metal-based NPs on crops, and the increased resistance ability of plants exposed to climatic stressor factors and pathogens, make nanotechnology a promising tool that is currently underused. Controversial results have been found in the published literature, which show positive or negative effects of NPs depending on many factors related not only to NPs' properties and plant species, but also to culture media and exposure conditions. The potential effects of NPs on plants due to the interaction with other contaminants have been less studied. The results confirm the active interactions between NPs and co-existing contaminants, which can be synergistic or antagonistic depending on the intrinsic properties of NPs and co-contaminants, plant species, and, more importantly, the application rate. Other factors like exposure mode, plant growth stage, and exposure time also influence joint toxicity. From a risk perspective, the occurrence of synergistic interactions is the biggest concern.

One of the most evident difficulties that limits the use of NPs in agriculture is to compare the results between the studies performed under different experimental conditions that determine outcomes. Therefore, a more systematic approach with standardized protocols that defines the many involved parameters as much as possible is necessary. In addition, a gap has been detected in knowledge of the real joint effects of NP-chemical mixtures. Further studies are needed to acquire more knowledge about the mechanisms of NP interactions with co-existing contaminants, including a comparative study with bulk particles and their ionic counterparts. The possible applications and uses of nanotechnologies in agriculture require the joint effects of NPs and co-contaminants being taken into account to establish regulatory guidelines.

Future research into metal-based NPs will address the precise release of nutrients adapted to soil features and crop needs. NPs will regulate the uptake of beneficial and harmful chemicals by plants. Simultaneously, NPs will allow plants to enforce their defenses against external stress agents and to improve their potential in stimulating plants to produce natural active molecules. Ultimately in the near future, NPs will enable us to accomplish sustainable agriculture by reducing inputs and chemical residues in crops.FundingThis chapter was supported by the Community of Madrid, project AGRISOST-S2018/BAA-4330.

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Chapter 3
Potential Toxic Effects of Metal or Metallic Nanoparticles in Plants and Their Detoxification Mechanisms

Zaid Ulhassan, Su Yang, Ali Raza Khan, Yasir Hamid, Sajid Muhammad, Wardah Azhar, Abdul Salam, Mohamed Salah Sheteiwy, Tariq Aftab, Ji An Wei, and Weijun Zhou

Abstract The development of nanotechnology via the enormous utilization of engineered nanoparticles (ENPs) in agriculture sector including metal (M) or metallic (MT) nanoparticles (NPs) has increased. Although, large surface area and particular sizes of NPs are some of the desired traits to ameliorate the environmental stress factors and modify the plant physiological processes, thereby leads to improve the crop production. The potential toxic nature of M or MT NPs (mainly depends upon their characteristics) sometimes causes toxic effects by inducing the extra production of reactive oxygen species, thus causing oxidative stress. Moreover, the M

Z. Ulhassan (✉) · A. R. Khan · S. Muhammad · W. Azhar · A. Salam · W. Zhou (✉)
Ministry of Agriculture and Rural Affairs Laboratory of Spectroscopy Sensing, Institute of Crop Science, Zhejiang University, Hangzhou, China
e-mail: wjzhou@zju.edu.cn

S. Yang
Ministry of Agriculture and Rural Affairs Laboratory of Spectroscopy Sensing, Institute of Crop Science, Zhejiang University, Hangzhou, China

College of Life Sciences, China Jiliang University, Hangzhou, China

Mizuda Group Co., Ltd, Huzhou, Zhejiang, China

Y. Hamid
Ministry of Education (MOE) Key Lab of Environmental Remediation and Ecosystem Health, College of Environmental and Resources Science, Zhejiang University, Hangzhou, China

M. S. Sheteiwy
Department of Agronomy, Faculty of Agriculture, Mansoura University, Mansoura, Egypt

T. Aftab
Plant Physiology Section, Department of Botany, Aligarh Muslim University, Aligarh, India

J. A. Wei
Mizuda Group Co., Ltd, Huzhou, Zhejiang, China

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or MT NP–induced oxidative stress impairs the cellular biomolecules and causes imbalance in plant biological and metabolic processes. In this way, it is crucial to explore the mechanisms involved in M or MT NP–mediated phytotoxicity along with NP detoxification strategies that can be helpful to understand the various morpho-physiological, biochemical, and metabolic responses of plants. To address this, recent studies concerning the M or MT NP toxicity at morpho-physiological, biochemical, cellular, and molecular levels have been discussed in this chapter. Factors affecting M or MT NP toxicity have also been overviewed. In addition, the potential detoxification strategies such as the applications of antioxidants, phytohormones, surface modifications, and omics approaches to minimize or ameliorate M or MT NP toxicity in different plants have been reviewed.

Keywords Nanoparticles · Nanotoxicity · Phytotoxicity · Oxidative stress · Antioxidants · Plants

3.1 Introduction

The higher concentrations of engineered nanoparticles (ENPs) such as metallic NPs within plant tissues has been a primary concern in health and safety issues because the dietary intake of these hazardous materials by food crops can pave the paths for human exposure. Despite popularity in agriculture sector as nano-fertilizer and pesticide delivery, their toxic impacts are still at the beginning stage. The distribution and transport of NPs within plant systems employed various unique pathways (Schwab et al., 2016). Different types of NPs displayed efficacy to protect plants from environmental stress factors which is enabled by different properties such as size and surface-to-volume ratio (Rajput et al., 2020; Nazir et al., 2022; Salam et al., 2022; Ulhassan et al., 2022a, b). These unique physio-chemical properties of NPs govern their toxic impacts (Shin et al., 2015). Due to small sizes, NPs can easily enter plant tissues, be transported to different parts, and cause damaging effects (Tripathi et al., 2017a). Most of the NPs caused phytotoxic effects (at particular concentrations) and disturbed the morpho-physiological, biochemical, and genetic characteristics (Rico et al., 2014; Tripathi et al., 2017a, b, c; Azhar et al., 2020; Khan et al., 2019, 2021). The key indicators of NP toxicity were the overproduction of reactive oxygen species (ROS), resulting in oxidative stress, peroxidation of lipids, and DNA and protein damages in different plants (Arruda et al., 2015; Ma et al., 2015; Khan et al., 2021). Although plants activate their inherent defense system to tackle the NP toxicity, sometimes these internal defense systems cannot cope with NP toxicity which leads to severe toxic impacts in plant cells (Tripathi et al., 2017a).

In this chapter, we have summarized the recent studies on the toxic effects of metallic NPs at morphological, physio-biochemical, anatomical, metabolic, cellular, and molecular levels. In addition, various strategies being utilized to detoxify or minimize the toxic effects of metallic NPs in plants have been discussed. These detoxification strategies include applications of phytohormones or signaling molecules, modification of adsorption surfaces, omics approaches, and endogenous production of antioxidant defense (enzymatic and non-enzymatic).

3.2 Toxic Effects of Metal-Based Nanoparticles in Plants

Depending upon the applied dose and particular nature of NPs, NPs can impose severe toxic effects in various plant species (Rastogi et al., 2017; Ulhassan et al., 2022a, b). Zinc oxide nanoparticles (ZnO NPs) are among the remarkably used NPs. An investigation confirmed that ZnO of 10–25 nm sizes and 500–1000 μg/ml caused inhibition of *B. oleracea* roots by 41% (Pokhrel & Dubey, 2013). The same level of ZnO NPs displayed severe toxic effects on *B. napus*, *C. sativus*, *L. sativa*, and perennial ryegrass. Diverse toxic impacts such as reduction in roots growth, broken root epidermis, and disturbances in cortical cells were noticed (Lin & Xing, 2008). Excessive ZnO NP levels (2000 mg/mL) retarded the pod development and yield of *Arachis hypogaea* (Prasad et al., 2012). The sizes 10–15 of ZnO NPs at 2 g/L decreased the photosynthetic pigment level, biomass production, and protein contents (Narendhran et al., 2016). Recent studies explored that ZnO NPs (300 and 500 mg/L) reduced the chlorophyll contents, stomatal closure, and oxidative stress-mediated damages to ultra-structural organelles in the tissues of *Arabidopsis thaliana* (Khan et al., 2019) and *O. sativa* (Khan et al., 2021). It has been found that ZnO NPs exhibited genotoxic and cytotoxic effects within root cells up to particular levels (Faizan et al., 2020). In terms of genotoxic effects, excessive ZnO NPs overproduced the reactive oxygen species. These type of stresses are inclined to govern DNA damages (Molnár et al., 2020). The ZnO NP concentrations at 0.2 g/L caused chromosomal aberration in 24 h and four-time reduction in mitotic index (MI) after 48 h of exposure in *Vicia faba* plants (Ghosh et al., 2016). The exposure of Zn NO_3 also possessed cytotoxic effects as proven by the biologically modified leaf extract of *Albizia saman* and their possible effects on *Drimia* (*D.) indica* root cells. A dose-dependent reduction in mitotic index was noticed in *D. indica* root tips (Daphedar & Taranath, 2018). These reports suggested that higher doses of ZnO NPs caused adverse effects (cytotoxic and genotoxic) on the overall plant growth performance.

The exogenous supply of silver nanoparticles (Ag NPs) was documented to facilitate distortion in cell wall and cause damages to root cell vacuoles (Mazumdar & Ahmed, 2011). The concentrations of Ag NPs at 40 mg/L (20 nm sizes) have severely reduced the rates of seed germination in three plant species and enhanced in one species (Yin et al., 2012). The structures of the primary roots of maize plants were distorted along with the reduction in plant biomass and transpiration rates (50–90%) at the exposure of citrate-Ag NP complexes (Hawthorne et al., 2012). Soil spiked with Ag NPs at 790 Ag/Kg significantly restricted the shoot and root biomass and root lengths of *Pinus muricata* seedlings (Sweet & Singleton, 2015). A recent investigation revealed that the exogenous application of 100 mg/L Ag NPs reduced the root length of *Arabidopsis thaliana* (Wang et al., 2020). Another study identified that soil spike with 365 Ag/Kg displayed a decline in shoot and root growth and obvious changes in root biomass in *H. vulgare* seedlings (González Linares et al., 2020). Regarding genotoxicity, Ag NPs release Ag^+ ions that greatly impart to DNA damages due to oxidative stress (Kruszewski et al., 2011). The treatments of 20 mg/L of biologically produced Ag NPs have enhanced the number of cell deaths in *A. cepa* roots. An increase in micro-nuclei (0.79%) (w.r.t applied doses) was confirmed after 6 h exposure time of Ag NPs along with DNA damages by COMET assay (Heikal et al., 2020). Even a low dose of Ag NPs caused genotoxic and cytotoxic effects in *A. cepa* plants. It has been reported that Ag NPs severely disturbed the cell membrane, ATP production, and replication of DNA (Tripathi et al., 2017b). The Ag NPs mediated enhancement in the overproduction of ROS (oxidative stress), lipid peroxidation, and DNA damages as already reported in recent studies (Rajput et al., 2020; Yan & Chen, 2019).

The elevated levels of copper oxide nanoparticles (CuO NPs) imposed toxic effects on plant growth and development. The numerous physiological processes were greatly impaired by the higher levels of CuO NPs which thereby overproduced the ROS (Azhar et al., 2020). In *Elodea densa*, the concentrations of 1–2 mg/L badly affected the photosynthesis (Nekrasova et al., 2011). A significant reduction in photosynthesis and increase in the activities of few antioxidant defense enzymes (ascorbate peroxidase and superoxide dismutase) was noticed in the leaves of *Oryza sativa* plants at 1000 mg/L Ag NPs (Da Costa & Sharma, 2016). In *C. sativum*, nano-CuO at 80 mg/Kg of dry soil caused more than 50% inhibition in seed germination and nutritional quality (Zuverza-Mena et al., 2015). A recent study reported the adverse effects of 50, 100, 200, 300, and 400 mg/L CuO NPs (37 nm sized) on the growth and biomass, chlorophyll contents, opening of leaf stomata, guard cells, and related sub-cellular structures (Azhar et al., 2020). In *A. thaliana*, the 30–50 size range of CuO NPs (10 mg/L) caused swelling of root hair zone, splatted cell wall, and scrambled cellular arrangements (Nie et al., 2020). Genotoxic effects in terms of DNA damages, methylation, and lipid peroxidation have been observed (Perreault et al., 2010). In *Fagopyum esculentum* (buckwheat), the exposure of CuO NPs at 2000–4000 mg/L caused DNA polymorphism (Lee et al., 2013).

Titanium oxide nanoparticles (TiO_2 NPs) have the potential to gather in roots and be transported via different channels. An investigation on TiO_2 NP colloidal suspension has been reported to reduce the leaf growth and water transpiration especially in root systems (Asli & Neumann, 2009). The elevated levels of TiO_2 NPs were found to restrict the nutritional quality of *C. sativum*. In addition, higher TiO_2 NP concentrations caused damages to root cellular membranes and growth performance of *C. sativum* (Hu et al., 2020). Overall, the elevated levels of TiO_2 NPs severely damaged the cell wall junctions and cellular arrangements. The 21 nm and 50 nm sizes of TiO_2 NPs (10, 100, and 1000 μg/L) mediated the genotoxic effects including damages to root meristem in *A. cepa*. The TiO_2 NPs facilitated DNA damages (as revealed by COMET assay) which were more pronounced at 21 nm (Demir et al., 2014). These alluring results were possibly associated with the different sizes of TiO_2 NPs that enable permeability of NPs (Fig. 3.1).

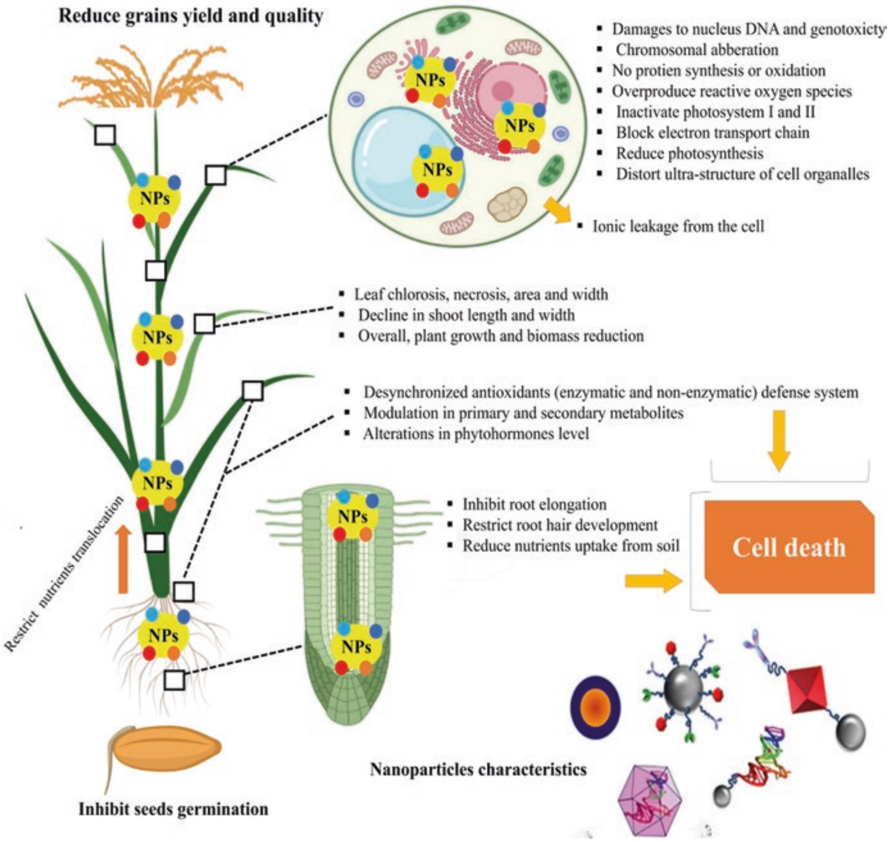

Fig. 3.1 Nanoparticle (NP)-induced phytotoxic effects are displayed at morpho-physiological, biochemical, and cellular levels

3.3 Factors Influencing the Toxic Effects of Nanoparticles

A number of factors are involved in modulating the NP-induced phytotoxic effects.

3.3.1 Form and Composition of NPs

The composition and type of NPs significantly affect their phytotoxicity in higher plants. It has been documented that 1000 mg/kg CeO_2 NPs disturbed the stress responses as revealed by the reduced levels of MDA and activities of SOD and POD enzymes in lettuce plants (Gui et al., 2015). In red spinach, the applications of multiwalled carbon nanotubes greatly enhanced the ROS generation in the vascular bundles and caused cell death due to necrosis (Begum & Fugetsu, 2012). The treatments of graphene family NMs resulted in the inhibition of seed germination, reduction in the water uptake by seeds, overproduction of H_2O_2, and related oxidative damages in the seeds of different vegetable crops such as cabbage, cucumber, and tomato. It also negatively affected the physio-biochemical levels such as metabolic modification, disturbances in antioxidant defense system, and extra generation of ROS (Wang et al., 2019). In addition, graphene oxide quantum dots as well as graphene oxides (reduced form) triggered the oxidative stress as observed by H_2O_2 and MDA levels in chili fruits. Also, the regulation of NPs uptake by plants is governed by the downregulation of trans-membrane transport proteins at the exposure of 0.25, 2.5 and 25 mg/kg FW in the stems of pepper (Li et al., 2005). It is clear that NPs imposed severe toxic effects in comparison to larger sizes of the same coating materials (Oberdürster, 2000).

3.3.2 *Difference in Seed Size or Quality and Plant Growth Stages*

To some extent, it was observed that the differences in seed sizes and xylem structures of multiple leaf plants also affected the toxic impacts of NPs (Lee et al., 2008). The responses of different plant seeds were tested against the same NPs. Three different rice varieties (high, medium, and low amylose) were grown in CeO NP–contaminated soils. It was noticed that the supply of CeO NPs enhanced the Ce contents in medium and low amylose treatments in comparison to their relative controls (Rico et al., 2013). As compared to conventional seeds, transgenic seeds responded differently (Gui et al., 2015). Salah et al. (2015) confirmed that the exposure of ZnO NPs reduced the surface area of rice cultivar (Zhu Liang You 06) as compared to Qian You No.1 under non-stressed conditions for both cultivars.

During the whole growth period, the photosynthetic parameters were altered in cucumber seedlings under 200 mg/L CeO NP treatments. In addition, the leaf area and size were decreased as compared to relative controls at seedling stage. But, mature leaves displayed no significant differences against the applied treatments (Hong et al., 2016). In *Phaseolus vulgaris*, the activities of CAT and APX enzymes were sharply upsurged in roots at 15 days than 7 days, except their relative controls. Total soluble protein showed opposite trends, except 500 mg/L CeO NP applications (Majumdar et al., 2014).

3.3.3 *Mode of NP Application and Applied Dose*

Plants respond differently to NP exposure, and their potential toxic effects may vary with the plant growth stage, mode of NP supply, time duration of NP exposure, and type of soil (Aslani et al., 2014; Ulhassan et al., 2022c). A recent study reported the possible toxic impacts of 10 or 250 mg per plant supply of CuO NPs (for 10 and 15 days) in *Brassica oleracea*. After 15 days of CuO NP exposure, lettuce and cabbage plants displayed higher Cu uptake and reduction in plant height, net photosynthesis rate, and water levels (Xiong et al., 2017). Energy dispersion X-ray microanalysis and scanning electron microscopy displayed deformation in stomata comprising CuO NPs (Raliya et al., 2016). There have been contradictory reports concerning the toxic impacts of metal-based NPs, and most of the studies found foliar spray as higher metal accumulator which caused less toxic effects in plants (Liu et al., 2017). Few studies found foliar spray of NPs as highly toxic and cause of higher accumulation of particles (Salehi et al., 2018), and others revealed lower particle accumulation and less phytotoxic effects (Li et al., 2020) as compared to NP supply via roots.

3.3.4 Coating Material

Surface coating plays crucial roles regarding their functions and utilization in soil and plant systems. These surfactants or coating materials enhance the stability of NPs via electrostatic repulsive forces among particles and provide and improve the hydrophobicity (Xu et al., 2012). In addition, surface characterizations govern the NP aggregation by modifying the surface charges and interactions with environmental factors (Chen, 2018). The stabilization of iron NPs with the surfactants such as coating of starch and polyacrylic acid provides electrosteric stabilization and protects NPs from the aggregation (Dong et al., 2013). The NP dissolution on surface coating and ligands (in media) is also affected by the organic coating of Ag NPs (Sharma et al., 2014). It has been documented that the foliar spray of uncoated NPs displayed has no effect on biomass production and photosynthesis in lettuce plants (Larue et al., 2014). In contrast, another study reported that the coating (with various materials) of Ag NPs enhanced the negative effects of Ag ions as observed by the plant-water relations (Torrent et al., 2020).

3.3.5 Application Media

A particular application media is obligatory for the delivery of NPs within plant cellular organs because NP characterization differs relying on application media. Toxic effects of NPs have been well-documented under diverse exposure media such as agar, nutrient solution, and aqueous suspension (Kouhi et al., 2015; Pokhrel & Dubey, 2013; Lee et al., 2012). To investigate the phytotoxic effects of NPs on lettuce plants, CeO NPs (1000 mg/kg) were directly applied on spotting soil, and it displayed negative effects on the plant growth traits, enhanced the levels of oxidative stress markers (overproduced ROS), elevated the production of MDA, and restricted the antioxidant defense system (Gui et al., 2015). The supplementation of

ZnO NPs leads to higher Zn accumulation by pea and beetroot plants. The application of 225 mg/kg of dry soil enhanced the ROS levels by 47% and 130% in both crops, respectively (García-Gómez et al., 2018). In recent investigation, the exposure of Ag NPs in Hoagland's nutrient solution displayed phytotoxic effects (especially at higher concentrations) on the growth of lettuce plants (Hasan et al., 2021).

3.4 Detoxification Mechanisms of Metal-Based Nanoparticles

Different strategies have been adopted to tackle the potential toxic effects of NPs in plants.

3.4.1 Antioxidants as ROS Scavenger

As the elevated levels of NPs overproduced the ROS in plant tissues, this is counteracted by antioxidants that generate harmless molecules such as hydrogen, water, and oxygen. In this way, antioxidants can be a better choice to scavenge the extra generation of ROS and their induced oxidative stress. Plants have efficient antioxidant defense system in the form of superoxide dismutase (SOD), peroxidase (POD), catalase (CAT) (Ulhassan et al., 2018, 2019b, c; Yang et al., 2021), glutathione reductase (GR) dehydro-ascorbate reductase (DHAR), monodehydroascorbate reductase (MDHAR) (Ulhassan et al., 2019a) and non-enzymatic antioxidants such as ascorbic acid, glutathione and phenolic compounds (Gill & Tuteja, 2010). The fabrication of these antioxidants with toxic NPs can be utilized to minimize the NP phytotoxicity. Earlier reports suggested that the supply of ascorbic acid with carbon nanotubes reduces the ROS production in red spinach (Begum & Fugetsu, 2012). The supplementation of fruits extract with ZnO NPs restored the glutathione activity (Singh et al., 2020) which suggested that fruit extract can be effective antioxidants to reduce NP toxicity and scavenge the ROS overproduction. In order to enhance the antioxidant capacity and minimize NP phytotoxicity, they can be modified to nanocomposites before their supply to plants. Anthocyanin is another non-enzymatic antioxidant that effectively scavenge the excessive production of free radicals and results in chelate formation with toxic metal ions (Carocho & Ferreira, 2013). Thus, the combined use of antioxidants and NPs had great potential to reduce NP phytotoxicity.

3.4.2 Modification of Adsorption Surfaces

The NP-induced toxic effects can be minimized by surface modifications of target NPs. Despite the same composition, different type of NPs have different toxic impacts, for instance, different forms of carbon NPs such as fullerenes, carbon fibers, carbon nanotubes, and carbon-graphite (Ge et al., 2012). A single-walled carbon nanotube distorts the mitochondria functions (Dong et al., 2013), and multi-walled carbon nanotube decreases the phagocytosis (Wang et al., 2012). These reports confirm that surface modification is a crucial step to reduce the NP-induced phytotoxicity.

3.4.3 Phytohormones or Signaling Molecules

The exogenous supplementation of phytohormones has emerged as an efficient strategy to reduce the NP (especially at higher concentrations)–mediated toxic effects. In wheat plants, authors reported the efficacy of gibberellic acid (GA) in minimizing the toxic effects induced by titanium dioxide (TiO_2) NPs. They found that the alone supply of 600 mg/kg TiO_2 NPs drastically reduced the plant growth and biomass, chlorophyll contents, and nutrient (Fe, P, Mn and K) uptake and impaired the antioxidant defense system as compared to their relative controls. The exogenous treatments of GA reduced the TiO_2 NP–facilitated phytotoxic effects as noticed by the elevated levels of nutrients, growth, yield, and antioxidant enzyme activities (SOD, CAT and APX) and reduction in oxidative stress (H_2O_2, MDA, and electrolyte leakage) (Alharby et al., 2021). Another recent study reported that the foliar spray of melatonin (ME) minimized the CuO NP and ZnO NP toxic impacts in the tissues of rice seedlings. It was observed that ME significantly alleviated the CuO NP and ZnO NP–induced oxidative stress, upsurged the transcript levels of genes involved in antioxidant defense, and enhanced the total amino acid content in the leaves of rice seedlings (Song et al., 2021). However, the synergistic effects of GA and ME with other NPs in different plants demand thorough understandings.

A few studies documented that the exogenous supply of phytohormones or signaling molecules can modulate or sometimes enhance NP toxicity in different plants. Azhar et al. (2020) documented that ethylene further triggered the CuO NP (particularly at 400 mg/L)–induced toxic effects in *Arabidopsis thaliana* leaves. They verified that 400 mg/L CuO NPs negatively affected the plant biomass, chlorophyll contents, stomatal aperture, guard cells, and other related cellular organelles. The insensitive mutants of ethylene displayed less susceptibility to cellular damages than wild-type seedlings. Moreover, the excessive CuO NPs upregulated the biosynthesis genes of ethylene. In contrast, the exogenous supply of ethylene biosynthesis antagonists (cobalt and silver ions) slightly enhanced the activities of antioxidant defense enzymes as compared to CuO NP stress alone treatments. Similarly, ethylene signaling pathways were involved in enhancing the ZnO NP toxicity in *Arabidopsis thaliana* leaves. In detail, ZnO NPs reduced the biomass, chlorophyll contents, and soluble sugar levels, overproduced the ROS, and regulated the cell-cycle genes. Higher transcript levels of antioxidant defense genes in mutant plants revealed their extra tolerance than wild-type plants in response to ZnO NP oxidative damages (Khan et al., 2019). A recent investigation revealed that exogenous supply of melatonin greatly reduced the CuO NPs (300 mg/L) induced toxic effects in maize seedlings. In details, melatonin enhanced the maize tolerance to CuO NPs toxicity, improved the plant growth performance, photosynthetic efficiency, antioxidants defense system and reduced the oxidative stress induced cellular damages (Khan et al., 2023). In another study, it was found that ethylene enhanced the CuO NPs (450 mg/L) induced toxic effects in rice seedlings. They documented that ethylene mediated the CuO NPs induce oxidative stress and cellular ultrastructural damages in plant tissues (Azhar et al., 2022). In green pea (*Pisum sativum Fabaceae*), the applications of indole-3-acetic acid regulated the toxic effects of CuO NPs. Their findings revealed that the soil amendments with 10 μM indole-3-acetic acid, 50 and 100 mg/kg CuO NPs (10–100 nm sizes), and 100 mg/kg $CuCl_2$ reduced the plant biomass by 50%. In contrast, a few combination of indole-3-acetic acid, particularly at 100 μM, altered the nutrient profiling, caused no effect on pod elements, notably enhanced the pod Fe and Ni, increased the Cu contents, and reduced Ca and Mg contents in stems. These outcomes suggested that CuO NPs have potential to enhance the nutritional value, but exogenous addition of indole-3-acetic acid can alter the pea production since these amendments diminished the pod biomass and reduced the number of plants (Ochoa et al., 2017).

3.4.4 Omics Approaches

Omics approaches such as genomics, transcriptomics, and proteomics have also been used (directly or indirectly) to reduce NP toxicity (Fröhlich, 2017). Among other uses of these techniques, a few uses include the identification of novel targets, stress extent, and molecular markers (protein or DNA-based) of NP toxicity (Patra et al., 2018). The tremendous progress in molecular biology tools such as next-generation RNA sequencing, DNA microarray, hybridization subtraction, serial analysis of gene expression (SAGE), and gene expression profiling has eased the identification of markers at any given time period. Proteomics analysis grants direct knowledge about the phenotype alterations because any cellular modifications by NP toxicity are related to the protein localization. Proteomics is a crucial tool to denote post-translational modification, protein-protein interactions, and regulation of proteins. In addition, protein levels can be determined in cells by two-dimensional gel electrophoresis, stable isotope labeling, fluorescence labeling, etc. (Liu et al., 2019). Metabolomics can be an important tool to identify NP toxicity by evaluating the alterations in cellular metabolites such as amino acids, lipids, carbohydrates, and vitamins. These metabolites can reflect the exact changes in cellular characterization (Mwamba et al., 2020).

Omics-based approaches can be utilized to identify the potential reason and solution of NP toxicity. The post-exposure of SiO_2, TiO_2, and ZnO NPs was identified by DNA methylation and hypomethylation state (Lu et al., 2016; Tabish et al., 2017). The metabolomics analysis explored the potential responses of cucumber plants to Cu-associated NP toxicity (Zhao et al., 2016). Gas chromatography-mass spectrometry and NMR spectroscopy techniques were adopted to find out the potential interaction of NP concentrations with the uptake of micro- and macro-nutrients that leads to metabolic changes in cucumber tissues. These markers may help scientists to overcome the causes of NPs toxicity. There is still a huge gap between the involved pathways and NP stress–linked mechanisms which should be explored to tackle NP toxicity. The comprehensive understandings of these omics approaches can be very effective to deal with NP toxicity (Table 3.1).

Table 3.1 The potential strategies utilized for the detoxification of NP-induced toxic effects in plants

Detoxification approaches	Plant species	Nanoparticles stress	Modulation in plant growth attributes	References
Antioxidants (enzymatic and non-enzymatic)	*Triticum aestivum*	5, 50, and 150 mg/L TiO_2 (21 nm size)	↑ Growth, enzymes of AsA-GSH cycle, total proteins, thiols ↓ H_2O_2 levels	Silva et al. (2019)
Elevated CO_2 levels	*Oryza sativa*	200 mg/kg ZnO (<100 nm size)	↑ Growth and biomass, grain protein, yield	Du et al. (2022)
Phytohormone (indole-3-acetic acid)	*Pisum sativum Fabaceae*	50 and 100 mg/kg CuO NPs (10–100 nm size)	↑ Growth, nutritional profiling, yield ↓ Pod biomass	Ochoa et al. (2017)
Phytohormone (melatonin)	*Oryza sativa*	50 and 500 mg/L ZnO NPs (30 ± 10 nm size)	↑ Growth, dry biomass, nutrient (Zn, Fe, and Mn) uptake, SOD, POD, and CAT activity ↓ Root ZnO NPs, MDA	Huang et al. (2020)
Gaseous signaling molecule (nitric oxide)	*Pisum sativum*	1000 and 3000 μM Ag NP (22 nm size)	↑ Growth, photosynthesis, SOD, GR, APX, and DHAR activity, total AsA and GSH, ↓ Ag uptake, H_2O_2 and MDA levels, cellular ultra-structural damages	Tripathi et al. (2017b)
Gaseous signaling molecule (nitric oxide)	*Triticum aestivum*	100 and 200 μM ZnO NPs (5–20 nm size)	↑ Growth and biomass, photosynthesis, enzymes of AsA-GSH cycle ↓ H_2O_2 and MDA levels	Tripathi et al. (2017c)
Gaseous signaling molecule (hydrogen sulfide)	*Oryza sativa*	100 μM CuO NPs (<50 nm size)	↑ Growth and biomass, photosynthetic pigments, antioxidant pool (SOD, APX, GR, and MDHAR) ↓ ROS (H_2O_2 and $O_2^{\bullet -}$), MDA	Rai et al. (2021)

(continued)

Table 3.1 (continued)

Detoxification approaches	Plant species	Nanoparticles stress	Modulation in plant growth attributes	References
Gaseous signaling molecule (hydrogen sulfide)	*Arabidopsis thaliana*	10 and 20 µg/L CuO NPs	↑ Growth attributes, antioxidant activity, GSH, cysteine cycle, ↓ Electrolyte leakage, ROS (H_2O_2), MDA levels, cellular damages	Jia et al. (2018)
Glutathione (GSH)	*Glycine max* (L.)	62.5 mg/kg Ag NPs	↑ Growth and fresh biomass, nitrogen content, thiol compounds, essential amino acids ↓ Ag accumulation, nodule number and weight	Ma et al. (2020)
Organic substance (humic acid)	*Oryza sativa*	500 and 750 mg/L ZnO NPs	↑ Seed germination, growth and biomass, proteins, starch and sugar contents ↓ Electrolyte leakage, genes involved in ABA synthesis, ROS (H_2O_2 and $O_2^{\bullet -}$), MDA, cell ultra-structure of leaf and root cells	Sheteiwy et al. (2017)

3.5 Conclusions and Future Perspectives

It is clear that the interaction of NPs with plants mainly depends upon the NP surface area, size, shape, chemistry of crystals, applied dose and seed quality, plant genotype, growth stage, applied medium (soil or water), and environmental conditions. The M- or MT-based NP toxicity occurs at all morphological, physiological, biochemical, anatomical, cellular, and molecular levels. The utilization of different NP detoxification strategies such as applications of antioxidants, phytohormones, or signaling molecules, surface modifications, and omics approaches provide clues for sustainable crop production. With the recent focuses on M or MT NP toxicity, researchers are trying to explore the entry point of NPs in plant cells and their distribution patterns that affect the plant growth and yield attributes, for instance, how these NPs pass through the cell membranes with the help of different transporters or carrier channels.

The understandings of signaling pathways of NPs and reactive oxygen species or reactive nitrogen species should be further understood. The dose- and time-based toxic effects of NPs should be carefully used in crop management. The extended exposure time period for NP dose can further strengthen the novel research allied to the accumulation and toxic effects of NPs in plants. The mechanisms involved in NP toxicity should be thoroughly explored (especially at cellular and molecular levels) before their field applications. More effective strategies should be focused on to handle NP toxicity. There are still rare studies (at whole plant life-cycle) on the molecular mechanisms of NP toxicity and their detoxification approaches under hydroponic and field conditions. These future research recommendations can be helpful to apprehend NP toxicity and adaptation of novel strategies to handle the nanotoxicology.

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Chapter 4
Non-standard Physiological Endpoints to Evaluate the Toxicity of Emerging Contaminants in Aquatic Plants: A Case Study on the Exposure of *Lemna minor* L. and *Spirodela polyrhiza* (L.) Schleid. to Dimethyl Phthalate (DMP)

Fabrizio Pietrini, Laura Passatore, Serena Carloni, and Massimo Zacchini

Abstract Emerging contaminants are targeted as primary pollutants of great concern for the ecosystem health. The evaluation of their effects on biota by non-destructive and reliable techniques is an issue receiving growing interest within the scientific community, particularly for plants. Beside commonly used proxies such as growth parameters and chlorophyll content, a set of non-standard endpoints has been proposed to be effectively used for the evaluation of the toxicity effects of xenobiotics on plants. Among them, chlorophyll fluorescence and leaf reflectance spectra analyses were reported as information-rich, non-destructive, and real-time mode technologies to assess the physiological status of plants. To verify their applicability for the ecotoxicology assessment in plants, a case study was set up by investigating the effects of dimethyl phthalate (DMP), an emerging contaminant in the aquatic environment, on two aquatic model plants (*Lemna minor* L. and *Spirodela polyrhiza* (L.) Schleid.), paralleling standard and non-standard toxicity endpoints. Results of a 7-day toxicity assay highlighted the effectiveness of chlorophyll fluorescence parameters and leaf reflectance spectra to assess the toxicity status of plants. Specifically, it was evidenced a lack of toxicity effects at 3 (DMP1) and 30 (DMP2) mg/L DMP in both *Lemna* and *Spirodela* plants, while at the highest DMP concentration (600 mg/L, DMP3), biometric and physiological endpoints revealed a toxicity status in both plant species, particularly in *Spirodela*. Thus, the performed case study evidenced the suitability of the targeted non-standard endpoints in

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F. Pietrini · L. Passatore · S. Carloni · M. Zacchini (✉)
Research Institute on Terrestrial Ecosystems (IRET), National Research Council of Italy (CNR), Monterotondo Scalo, Rome, Italy
e-mail: massimo.zacchini@cnr.it

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assessing the ecotoxicity of DMP in aquatic plants, also contributing to shed light on the environmental risk posed by phthalates in the freshwater compartment.

Keywords Chlorophyll fluorescence · Duckweed · Ecotoxicity · Leaf reflectance spectra · Pigments · Xenobiotics

4.1 Introduction

The widespread occurrence of chemical compounds from anthropogenic sources is currently representing a severe concern at global scale. Among such compounds, in recent years, a specific attention has been devoted to a heterogeneous group of substances, ranging from pharmaceuticals and personal care products to plastic additives and flame retardant, overall termed as "compounds of emerging concern (CEC)". These substances are generally characterised for a lack of regulatory standard also due to the fragmentary information on their toxicity threshold in the different environmental matrices. In this regard, the evaluation of the toxicological risk associated with the presence of emerging contaminants is considered a relevant issue, particular for the aquatic ecosystem (Riva et al., 2019; Tang et al., 2019). In fact, due to the vulnerability of the complex ecological network characterising this compartment, freshwater ecosystem is considered greatly exposed to chemical perturbations. In order to monitor the quality of the water, biological organisms are long proposed and utilised both at animal and plant levels.

As representative of aquatic macrophytes able to indicate the biological quality of the freshwater ecosystem, duckweed species (Lemnoideae) are largely studied for their responses to environmental pollutants (Radić et al., 2011; Baudo et al., 2015; Pietrini et al., 2015; Forni & Tommasi, 2016; Ziegler et al., 2019), being officially utilised for ecotoxicological assay (OECD n. 221, 2006).

Among endpoints targeted to assess the toxicity effects of chemicals on duckweed plants, growth and chlorophyll content are commonly investigated. However, as pointed out by Ziegler et al. (2019), such traits are very useful to highlight an overall toxicity of a certain water contaminant, but they bring very poor information about the toxicity mechanisms underlying the observed effect. In this regard, a set of non-standard endpoints associated with physiological, biochemical, and molecular mechanisms has been proposed (Mkandawire et al., 2014; Basiglini et al., 2018; Ziegler et al., 2019). The effectiveness of the investigated traits in assessing the toxicity status of duckweed plants has been proved under multiple trials. Anyway, a drawback of this approach is related to the destructivity mode of the performed analysis. Recently, to make a step forward, an experimental set-up aimed at evaluating biometric and photosynthetic traits in real time and in a non-destructive way in aquatic plants has been designed (Pietrini & Zacchini, 2020). Such device, termed the Eco-Tox Photo system Tool (ETPT), is capable to get an insight on the physiological state of photosynthetic apparatus, providing larger number of indices associated with toxicity endpoints linked to plant biometric, physiology, and biochemistry, on the same plant material and in the same time interval. Specifically, an integrated instrumental approach allows obtaining simultaneous information on the physiological and biochemical traits associated with a toxicity status by analysing the leaf chlorophyll fluorescence parameters, the biometric indices, and the leaf reflectance spectra (Pietrini et al., 2019, 2022). With the aim to further investigate on the suitability of such approach in describing the toxicity effects of a contaminant in aquatic plants and possibly validate it by comparison with standard endpoints, a case study was set-up by exposing *S. polyrhiza* and *L. minor* plants to dimethyl phthalate (DMP), as representative of phthalates, a very concerned class of xenobiotics for the freshwater ecosystem. To this scope, a 7-day test was performed to study growth and physiological responses, by analysing standard and non-standard parameters, in plants exposed to different concentrations of DMP chosen among those used for toxicity assessment of phthalates in plants (Kumari & Kaur, 2020).

The suitability of using duckweed species to investigate on the effects of emerging contaminants to aquatic biota has been largely observed. In this regard, Di Baccio et al. (2017) and Pietrini et al. (2019) reported on the effects of ibuprofen and perfluorooctanoic acid (PFOA) in *Lemna minor* L., respectively. Moreover, Hájková et al. (2019) highlighted the negative impacts of diclofenac on the photosynthetic process in *L. minor*. Singh et al. (2018) described the phytotoxicity of the antibiotic amoxicillin to *Spirodela polyrhiza*, and Alkimin et al. (2019) studied the effects of different pharmaceuticals on non-standard endpoints in *L. minor* and *L. gibba*.

In the last few years, phthalates (PAEs, dialkyl or alkyl aryl esters of 1,2-benzenedicarboxylic acid) have been recognised among the emerging contaminants of increasing concern for the ecosystem and human health. Their large worldwide utilisation, both as plasticisers and in personal care products, has rapidly enhanced the presence of phthalates in the environmental matrices (Fromme et al., 2002). This aspect, together with the well-documented adverse effects on humans, ranging from diabetes to pregnancy loss (Kumari & Kaur, 2020), and toxicities at reproductive and developmental levels in other animals (Yost et al., 2019), has forced the EU Commission to put phthalates under regulation. Particularly, the presence of PAEs in the plastic products used in agriculture (irrigation tubing and film mulching, among others) has been considered as a potential risk for their transfer to soil and irrigation water, and possibly for the accumulation into vegetables, leading to their enrichment through the trophic chain to humans (Ning et al., 2017). In fact, due to their lipophilic nature, a notable accumulation and translocation of PAEs in cultivated plants have been reported (Sun et al., 2015; 2018a). Under laboratory trials, the presence of different PAEs in the growth medium has been reported to cause adverse effects in plants at morphological, physiological, and biochemical levels (Kumari & Kaur, 2020). Negative effects caused by the exposure to PAEs were reported for plant biomass production (Liao et al., 2006), shoot and root elongation (Ma et al., 2013), and seed germination (Zhang et al., 2016). Phthalates were claimed to modify the oxidative status of plant cells by generating reactive oxygen species (ROS) and altering the activities of antioxidative enzymes (Ma et al., 2014), thus resulting in membrane lipid peroxidation (Zhang et al., 2015; Gao et al., 2019). At the physiological level, effects on plant pigment content (Zhang et al., 2016) and on the photosynthetic performances (Gao et al., 2018) have been observed in PAE-treated plants.

Toxicity effects of PAEs on duckweed plants have been described by several authors, both at physiological and genomic levels (Pietrini et al., 2022). *Spirodela polyrhiza* plants showed different levels of growth inhibition after treatment with diethyl phthalate (DEP), benzyl butyl phthalate (BBP), di-n-butyl phthalate (DBP), and diallyl phthalate (DAP) (Cheng & Cheng, 2012; Kaur et al., 2017; Sharma & Kaur, 2019). The alteration of the pigment content in *Spirodela* plants caused by the exposure to DBP (Huang et al., 2006), di-2-ethylhexyl phthalate (DEHP) (Xu et al., 2010), and DEP (Cheng & Cheng, 2012; Sharma & Kaur, 2020) was also reported. At the biochemical level, the modifications of the oxidative stress markers (i.e. MDA and H_2O_2 content) in *Spirodela polyrhiza* plants after DEP treatment were observed by Cheng and Cheng (2012) and in *Spirodela polyrhiza* and *Lemna minor* exposed to DBP and DEHP by Huang et al. (2006) and Xu et al. (2010). Moreover, an increase in the activities of antioxidative enzymes in *Spirodela polyrhiza* plants treated with DAP and DBP was reported by Sharma and Kaur (2019) and Huang et al. (2006), respectively, while a decline in SOD and POD and an increase in APX, CAT, and GR was shown by Cheng and Cheng (2012).

In this work, a case study has been designed to evaluate the suitability of non-standard endpoints, in comparison to standard ones, to describe the toxic effects of DMP in duckweeds. Results of this study could contribute to obtain a more comprehensive ecotoxicological assessment in aquatic plants by suggesting the addition of the most fitting non-standard physiological endpoints in the battery of monitored proxies.

4.2 Experimental Approach

The case study is referred to a laboratory trial aimed at evaluating the toxicity effects of DMP on duckweeds through the analysis of standard and non-standard endpoints. Specifically, a 7-day assay, designed taking into consideration the guidelines for ecotoxicological test (OECD 221 guidelines, 2006), was set-up by exposing two duckweeds species, i.e. *Lemna minor* L. and *Spirodela polyrhiza* (L.) Schleid., to three different DMP concentrations among those commonly reported in literature for similar investigations. Fronds of the two plant species, collected by the plant maintenance stocks at IRET-CNR (Pietrini et al., 2019), were subjected to a disinfection step consisting of a gentle rinsing in a 0.5% bleach solution for 1 min followed by a multiple washing with distilled sterilised water. Successively, 2 fronds of *Lemna* were placed in a well (24-well sterile plates, TPP, Trasadingen, Switzerland) containing 2 mL half-strength Hoagland solution, while 3 fronds of

Spirodela were placed in a well (12-well sterile plates, TPP, Trasadingen, Switzerland) containing 5 mL one-fifth strength Hoagland solution, under a laminar flow hood to minimise the contamination. The two diluted Hoagland's nutrient solutions were used as control condition (*C*) to which DMP was added to reach three nominal concentrations (DMP1, 3 mg/L; DMP2, 30 mg/L; DMP3, 600 mg/L). DMP solutions were prepared by diluting the DMP product purchased from Sigma-Aldrich (St. Louis, MO, USA) in Hoagland's nutrient solution. The nutrient solution was sterilised by autoclave (121 °C for 25 min), and the DMP solutions were added by filtration (22 μm) after autoclaving under a laminar flow hood. Polystyrene plates used in this trial were free of plasticizers or other chemical additives, as stated by the producing company (TPP, Trasadingen, Switzerland). The covered plates were then placed on a rotary plate under gentle rotation (50 rpm) in a growth chamber with a temperature of 25 °C, a photoperiod 16 h light/8 h dark, and an irradiance of 60 μmol m^{-2} s^{-1}. To assess the morphological and physiological endpoints, both standard and non-standard ones, in order to evaluate the effects of DMP on duckweeds at the end of the 7-day assay, an experimental device termed Eco-Tox Photo system Tool (ETPT) was used (Pietrini & Zacchini, 2020). Such device, set-up in our laboratory, is an integrated instrumental approach constituted by three different instrumentations devoted to assess the physiological condition of the plant photosynthetic apparatus through the measure of leaf gas exchange (HCM 1000,Walz, Germany), images of leaf chlorophyll fluorescence (Imaging PAM, Walz, Germany), and leaf reflectance spectra (ASD FieldSpec-3 spectroradiometer (Analytical Spectral Devices Inc., Boulder, Colorado, USA). Such instruments are associated with a sealed transparent plexiglass chamber in which the multi-well plates are placed for the measurement procedure. Through the near infrared (NIR) imaging capture of plants by the Imaging PAM camera and the following image analysis by a specific software (ImageJ, IJ 1.46r, http://imagej.nih.gov/ij/), a set of biometric parameters is obtained (Di Baccio et al., 2017). By using the ETPT, therefore, it is possible to estimate in real time and in a non-destructive way the alteration of biometric and physiological endpoints, both standard as growth parameters and chlorophyll content and non-standard as chlorophyll fluorescence parameters and values of spectral indices.

Specifically, at the beginning (t_0) and at the end of the experimental trial (7 days, 168 h, t_7), the following biometric endpoints were analysed on *Lemna* and *Spirodela* plants, as reported by Pietrini et al. (2019):

- Mean frond area at the end of the experiment (MFA)
- Total frond number (FN)
- Multiplication rate, calculated on the basis of changes in FN (MR)
- Doubling time of frond number (T_d)
- Average specific growth rate, calculated on the basis of changes in FN, $\mu_{(t0-7)}$

Moreover, the total chlorophyll content, as standard endpoint, was calculated by applying the modified equation of Evans (1993) as reported by Pietrini et al. (2019).

About chlorophyll fluorescence, the following non-standard endpoints were measured:

- Maximum quantum yield of photosystem II (PSII) photochemistry (F_v/F_m)
- Effective PSII quantum yield (Φ_{II})
- Photosynthetic electron transport rate through PSII (ETR)
- Quantum yield of regulated (Φ_{NPQ}) and non-regulated (Φ_{NO}) energy dissipation in PSII
- PAR-absorptivity (Abs) apparent absorptivity of the leaf surface

The measurements were performed on 30 min dark-adapted fronds (F_v/F_m) and on fronds adapted to a light intensity of 60 μmol m^{-2} s^{-1} for at least 10 min to reach a steady-state condition (Φ_{II}, Φ_{NPQ}, Φ_{NO}, and ETR). The abovementioned parameters were calculated as reported by Di Baccio et al. (2017) and Kramer et al. (2004).

Furthermore, to provide detailed information on the saturation characteristics of electron transport, as well as the overall photosynthetic performance of *Lemna* and *Spirodela* plants, the light response curves (LCs) of electron transport rate (ETR) were obtained exposing the plants to five increasing irradiance levels, from 60 to 410 μmol m^{-2} s^{-1}. By fitting the LCs to the equation of Platt et al. (1980), the following parameters were calculated:

- Photosynthetic efficiency (α)
- Maximum electron transport rate (ETR_{max})
- Minimum saturating irradiance (E_k) calculated as (ETR_{max}/α)

Leaf reflectance spectra were acquired in the spectral range of 350–1025 nm. Data were sampled at intervals of 1.4 nm with a spectral resolution of 3 nm, and a white reference panel (Spectralon) was used to calibrate spectral measurement. Moreover, an external 50 W halogen lamp was set up with an illumination angle of 45°. Specifically, the gun probe was placed on a stand unit orthogonally to the plate, carefully respecting a fixed distance (0.5 cm) to the plant layer in order assuring a field of view (FOV) of approximately 2 mm (using a 25° bare fibre-optic). Reflectance spectra were recorded as the ratio of sample data to white reference (99% reflectance Spectralon panel) data under the same illumination and viewing conditions. The mean of the five spectra was then determined to provide a single spectral value. Six spectral reflectance indices, photochemical reflectance index (PRI); pigment specific simple ratio (PSSR) for Chl_a ($PSSR_a$), Chl_b ($PSSR_b$), and carotenoids ($PSSR_c$); normalised phaeophytinization index (NPQI); and anthocyanin reflectance index (ARI), were derived from the collected data and calculated according to the following equations, where *R* is the reflectance value measured in each band expressed in nm that is indicated by the subscript number:

- Photochemical reflectance index (PRI) = $(R_{531} - \mathrm{R}_{570})/(R_{531} + \mathrm{R}_{570})$ (Gamon et al., 1997)
- Pigment specific simple ratio a ($PSSR_a$) = $(R_{800})/(R_{680})$ (Blackburn, 1998)
- Pigment specific simple ratio b ($PSSR_b$) = $(R_{800})/(R_{635})$ (Blackburn, 1998)
- Pigment specific simple ratio c ($PSSR_c$) = $(R_{800})/(R_{470})$ (Blackburn, 1998)
- Normalised phaeophytinization index (NPQI) = $(R_{415} - R_{435})/(R_{415} + R_{435})$ (Peñuelas et al., 1995)
- Anthocyanin reflectance index (ARI) = $(1/R_{550})/(1/R_{700})$ (Gitelson et al., 2001)

Data obtained and shown in tables and figures of this chapter are referred to a trial set-up in duplicate, with 8 replicates (each corresponding to a single well in the multi-well plates) for each treatment for both plant species ($n = 8$), unless otherwise stated. One-way ANOVA was used to process normally distributed data in order to evaluate the effects of the different DMP concentrations on each plant species, using the SPSS (Chicago, IL, USA) software tool. Statistical significance of the mean data was assessed by Tukey's test ($P \leq 0.05$), unless otherwise stated.

4.3 Results and Discussion

As valuable standard endpoints for performing toxicity assessment in duckweeds (Radić et al., 2011; Di Baccio et al., 2017; Pietrini et al., 2019), biometric indices are commonly used for evaluating the effects of a suspected toxicant on plant growth processes. Biometric indices are also recommended in the guidelines for ecotoxicity assay by official protocols in duckweeds (ISO 20079, 2005; OECD 221, 2006). In the present case study, biometric indices, evaluated at the end of the DMP treatments (168 h/7 days), revealed that the growth of *Lemna* and *Spirodela* plants was substantially unaffected by the exposure to the lowest DMP concentrations tested. On the contrary, after the treatment with 600 mg/L DMP, the main growth parameter values were reduced in both plant species, revealing a toxic effect (Tables 4.1 and 4.2). Specifically, in *Lemna*, the highest DMP concentration caused an inhibition of approximately 70% of MR, $\mu_{(t0-7)}$, and FN and 60% of T_d, with respect to control (Table 4.1), while in *Spirodela* plants such treatment reduced approximately by 80% the MR and $\mu_{(t0-7)}$ and 60% of FN and T_d in comparison to control (Table 4.2). A similar feature was observed during the time-course of a previous work targeting DMP exposure in duckweeds (Pietrini et al., 2022). While no studies dealing with

Table 4.1 Growth parameters of *Lemna minor* L. treated with different DMP concentrations for 168 h/7 days

Parameter	*C*	DMP1	DMP2	DMP3
MFA (mm^2)	7.78 ± 0.14 a	8.16 ± 0.19 a	8.07 ± 0.17 a	5.84 ± 0.26 b
FN	11.3 ± 0.3 a	10.5 ± 0.3 ab	10.2 ± 0.3 b	3.6 ± 0.1 c
MR	104.5 ± 2.3 a	102.5 ± 2.2 a	101.1 ± 2.1 a	31.5 ± 3.1 b
T_d	2.11 ± 0.02 b	2.13 ± 0.02 b	2.15 ± 0.02 b	3.36 ± 0.11 a
$\mu_{(t0-7)}$	0.240 ± 0.005 a	0.236 ± 0.005 a	0.232 ± 0.005 a	0.072 ± 007 b

Treatments are *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

MFA mean frond area at the end of the experiment, *FN* total frond number, *MR* multiplication rate, calculated on the basis of changes in FN, T_d doubling time of frond number, $\mu_{t(0-7)}$ average specific growth rate, calculated on the basis of changes in FN

Table 4.2 Growth parameters of *Spirodela polyrhiza* L. treated with different DMP concentrations for 168 h/7 days

Parameter	*C*	DMP1	DMP2	DMP3
MFA (mm^2)	23.73 ± 0.76 a	23.45 ± 0.96 a	23.31 ± 0.51 a	24.86 ± 1.11 a
FN	9.6 ± 0.5 a	9.6 ± 0.5 a	10.1 ± 0.1 a	3.9 ± 0.1 b
MR	74.7 ± 2.4 a	71.6 ± 3.6 a	68.7 ± 2.9 a	15.1 ± 2.1 b
T_d	2.45 ± 0.03 b	2.50 ± 0.05 b	2.54 ± 0.04 b	4.23 ± 0.30 a
$\mu_{(t0-7)}$	0.172 ± 0.005 a	0.164 ± 0.008 a	0.158 ± 0.006 a	0.034 ± 004 b

Treatments are *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

MFA mean frond area at the end of the experiment, *FN* total frond number, *MR* multiplication rate, calculated on the basis of changes in FN, T_d doubling time of frond number, $\mu_{t(0-7)}$ average specific growth rate, calculated on the basis of changes in FN

Lemna exposure to PAEs are present in the literature, only few works are targeting *Spirodela* plants. According to the results of the present study, Cheng and Cheng (2012) observed a decline of the growth associated with increased levels of DEP in a similar 7-day test. Moreover, a biomass reduction in *Spirodela* plants exposed to increasing concentrations of BBP and DBP and DAP was reported by Kaur et al. (2017) and Sharma and Kaur (2019), respectively. A reduction in the length of *Cucumis* seedlings as a consequence of the treatment with DMP concentrations over 50 mg/L was reported by Zhang et al. (2016).

At physiological level, the official ecotoxicological assays set up for plants indicate the chlorophyll content as valuable proxy to evaluate the effects of a suspected toxicant on a plant primary process such as photosynthesis. Beside this standard endpoint, other physiological traits have been proposed as suitable indicators of the impairment status of the photosynthetic machinery (Dewez et al., 2018; Pietrini et al., 2019). Among them, non-standard endpoints have been targeted by several authors (Dong et al., 2002; Iriel et al., 2015; Dewez et al., 2018; Alkimin et al., 2019; Pietrini et al., 2016, 2019; Mateos-Cárdenas et al., 2019) by investigating the effects of xenobiotics on chlorophyll fluorescence parameters and spectral reflectance indices in duckweeds fronds. In the present case study, therefore, the analyses of chlorophyll content were paralleled by the acquisition of chlorophyll fluorescence data and imaging and spectral reflectance values by the ETPT equipped with the ASD FieldSpec-3 spectroradiometer (see Sect. 4.2). In Tables 4.3 and 4.4, values of the chlorophyll fluorescence parameters (F_v/F_m, Φ_{II}, Φ_{NPQ}, Φ_{NO}, and ETR), chlorophyll content, and PAR-absorptivity (Abs) for *Lemna* and *Spirodela*, respectively, are reported. Similarly to the biometric indices, only the highest DMP concentration (DMP3) significantly affected the abovementioned parameters in both plant species except for F_v/F_m and Φ_{NO} in *Lemna minor*. In particular, DMP3 treatment caused, in *Lemna* plants, a reduction of approximately 21% of Φ_{II}, 24% of ETR, 20% of chlorophyll content, and 4% of PAR-absorptivity (Abs) and an

Table 4.3 Chlorophyll content (Tot Chl), PAR-absorptivity (Abs) and chlorophyll fluorescence parameters, maximum quantum yield of PSII photochemistry (F_v/F_m) measured in dark-adapted fronds, and quantum efficiency of PSII photochemistry (Φ_{II}), non-photochemical quenching (Φ_{NPQ}) and (Φ_{NO}), and electron transport rate (ETR) measured at steady state with light intensity of 60 μmol photons m^{-2} s^{-1} in fronds of *Lemna minor* L. treated with different DMP concentrations (*C*, 0 μg/L (control), DMP1, 3 μg/L; DMP2, 30 μg/L; DMP3, 600 μg/L) for 7 days

Treatment		Chlorophyll fluorescence parameters and total chlorophyll content					
	F_v/F_m (rel. un.)	Φ_{II} (rel. un.)	Φ_{NPQ} (rel. un.)	Φ_{NO} (rel. un.)	ETR (μmol elect m^{-2} s^{-1})	Abs (rel. un.)	Tot Chl (g m^{-2})
C	0.791 ± 0.007 a	0.580 ± 0.014 a	0.124 ± 0.009 b	0.294 ± 0.011 a	12.29 ± 0.31 a	0.846 ± 0.005 a	0.372 ± 0.018 a
DMP1	0.793 ± 0.010 a	0.597 ± 0.016 a	0.123 ± 0.010 b	0.278 ± 0.012 a	12.60 ± 0.35 a	0.844 ± 0.004 a	0.362 ± 0.010 a
DMP2	0.789 ± 0.016 a	0.582 ± 0.026 a	0.135 ± 0.016 b	0.282 ± 0.011 a	12.29 ± 0.57 a	0.844 ± 0.006 a	0.365 ± 0.017 a
DMP3	0.775 ± 0.007 a	0.456 ± 0.023 b	0.227 ± 0.019 a	0.317 ± 0.013 a	9.29 ± 0.47 b	0.815 ± 0.008 b	0.299 ± 0.015 b

Data are the mean values of 8 replicates ± standard error (SE). A one-way analysis was applied, and data followed by different letters in the same column are significantly different (Tukey's test, $P \leq 0.05$)

Table 4.4 Chlorophyll content (Tot Chl), PAR-absorptivity (Abs) and chlorophyll fluorescence parameters, maximum quantum yield of PSII photochemistry (F_v/F_m) measured in dark-adapted fronds, and quantum efficiency of PSII photochemistry (Φ_{II}), non-photochemical quenching (Φ_{NPQ}) and (Φ_{NO}), and electron transport rate (ETR) measured at steady state with light intensity of 60 µmol photons m^{-2} s^{-1} in fronds of *Spirodela polyrhiza* L. treated with different DMP concentrations (*C*, 0 µg/L (control), DMP1, 3 µg/L; DMP2, 30 µg/L; DMP3, 600 µg/L) for 7 days

	Chlorophyll fluorescence parameters and total chlorophyll content						
Treatment	F_v/F_m (rel. un.)	Φ_{II} (rel. un.)	Φ_{NPQ} (rel. un.)	Φ_{NO} (rel. un.)	ETR (µmol elect m^{-2} s^{-1})	Abs (rel. un.)	Tot Chl (g m^{-2})
C	0.792 ± 0.001 a	0.592 ± 0.011 a	0.137 ± 0.010 b	0.263 ± 0.005 b	12.80 ± 0.34 a	0.858 ± 0.003 a	0.406 ± 0.010 a
DMP1	0.792 ± 0.002 a	0.573 ± 0.007 a	0.152 ± 0.008 b	0.271 ± 0.004 b	12.30 ± 0.22 a	0.852 ± 0.003 a	0.385 ± 0.011 a
DMP2	0.793 ± 0.002 a	0.586 ± 0.008 a	0.130 ± 0.005 b	0.272 ± 0.001 b	12.66 ± 0.24 a	0.853 ± 0.002 a	0.389 ± 0.008 a
DMP3	0.705 ± 0.007 b	0.452 ± 0.023 b	0.206 ± 0.021 a	0.318 ± 0.009 a	8.78 ± 0.49 b	0.775 ± 0.011 b	0.235 ± 0.015 b

Data are the mean values of 8 replicates ± standard error (SE). A one-way analysis was applied, and data followed by different letters in the same column are significantly different (Tukey's test, $P \leq 0.05$)

increase of about 45% of Φ_{NPQ} compared to control (Table 4.3). At the same time, in *Spirodela* plants, DMP3 treatment affected to a higher extent the physiological parameters, decreasing approximately by 24% the Φ_{II}, 31% the ETR, 42% the chlorophyll content, 10% the PAR-absorptivity (Abs), and the F_v/F_m ratio and increasing by 33% the Φ_{NPQ} and 17% the Φ_{NO} values with respect to control (Table 4.4). The physiological responses of the two plant species to DMP3 differed mainly in two chlorophyll fluorescence parameters, namely, F_v/F_m and Φ_{NO} (Tables 4.3 and 4.4). In particular, the significant reduction in the F_v/F_m ratio, only detected in *Spirodela* plants, may be attributable to a damage or inactivation of PSII reaction centres with subsequent photoinhibition of photosynthesis (Guidi et al., 2019). Moreover, in the same treatment condition, the significant increase in the Φ_{NO} values, also in this case only observed in *Spirodela* plants, indicates that both protective regulatory mechanisms and photochemical energy conversion are inefficient (Sánchez-Moreiras et al., 2020).

To get further insight on the physiological endpoints able to highlight the toxic effects of DMP on duckweeds plants, the analysis of the chlorophyll fluorescence images was performed to provide information on the spatial heterogeneity of photosynthetic performance of duckweed fronds. Representative images of the chlorophyll fluorescence parameters (F_v/F_m, Φ_{II}, Φ_{NPQ}, and Φ_{NO}) in fronds of *Lemna* and *Spirodela* are reported in Figs. 4.1 and 4.2, respectively. It is noteworthy that parameters measured in both species in control, DMP1, and DMP2 treatments and in *Lemna* plants exposed to DMP3 revealed an almost homogeneous pattern of distribution of chlorophyll fluorescence (Figs. 4.1 and 4.2). Contrarily, they showed a notable heterogeneous pattern of light utilisation and photosynthetic activity in *Spirodela* plants exposed to DMP3, with large necrotic areas observed (Fig. 4.2). As mentioned before, values of the standard endpoints as biometric parameters and chlorophyll content paralleled non-standard ones as PAR-absorptivity (Abs) and chlorophyll fluorescence parameters, highlighting a stress condition more pronounced in *Spirodela* plants when exposed to DMP3 (Tables 4.3 and 4.4). In this regard, Sharma and Kaur (2020) reported a reduction of the photosynthetic pigment content in *S. polyrhiza* treated with DEP, while the same feature was observed by Kaur et al. (2017), after exposure to DBP and BBP, and by Huang et al. (2006) and Xu et al. (2010) with DBP and DEHP treatments, respectively. Interestingly, a D1 protein degradation was evidenced by Cheng et al. (2013) in DEP-exposed *S. polyrhiza* plants, arguing for the adverse effect of this phthalate compound on the PSII activity.

In plants exposed to toxic compounds, parameters useful to evaluate the photosynthetic performance plants under different light regimes can be obtained also by analysing the light response curves (LCs) of electron transport rate (ETR). In the present case study, results of such analyses are reported in Tables 4.5 and 4.6 and Figs. 4.3 and 4.4. Data obtained confirmed the results observed in the above reported physiological measurements (Tables 4.3 and 4.4). In fact, the pattern of LCs was not affected by exposure to DMP at the lower concentrations (DMP1 and DMP2) for both plant species as confirmed by unchanged values of photosynthetic efficiency

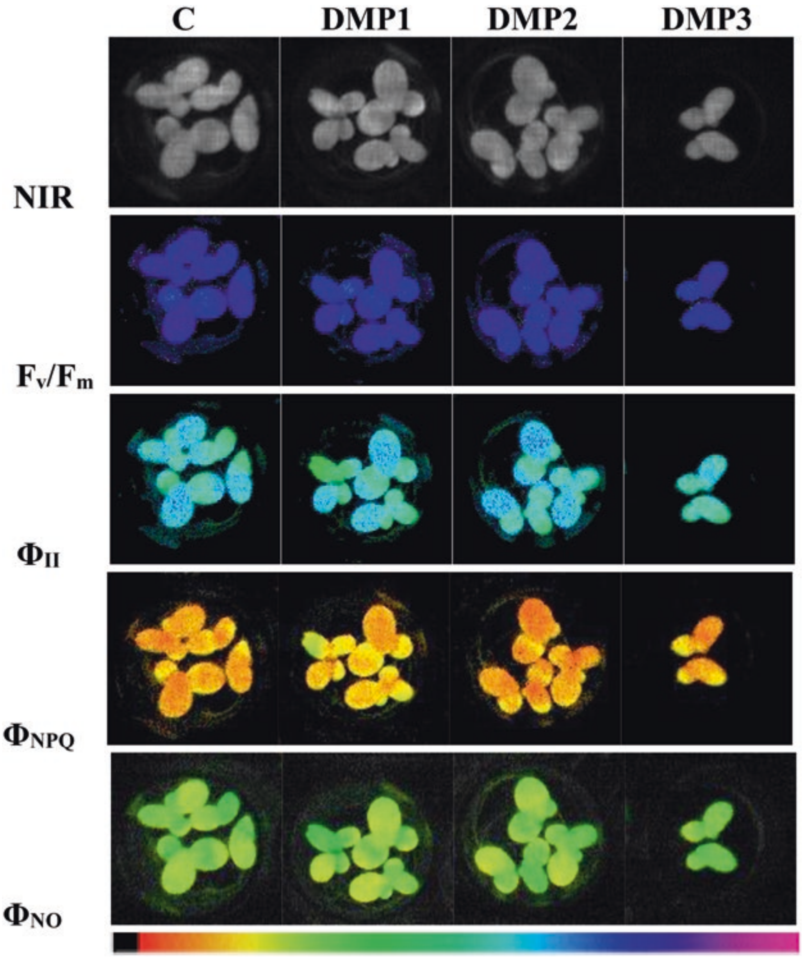

Fig. 4.1 Near infrared (NIR) and chlorophyll fluorescence images of maximum quantum yield of PSII photochemistry (F_v/F_m) in dark-adapted fronds and the effective PSII quantum yield (Φ_{II}) and the quantum yield of regulated (Φ_{NPQ}) and non-regulated (Φ_{NO}) energy dissipation in PSII at steady-state with actinic illumination of 60 μmol photons m^{-2} s^{-1} measured at the end of the experiment (7 days) in fronds of *Lemna minor* L. treated with different DMP concentrations (*C*, 0 mg/L (control), DMP1, 3 mg/L; DMP2, 30 mg/L; DMP3, 600 mg/L). The false colour code depicted at the bottom of the images ranges from 0.000 (black) to 1.000 (pink)

(α), maximum electron transport rate (ETR_{max}), and the minimum saturating irradiance (E_k) (Tables 4.5 and 4.6). Conversely, the highest DMP concentration (DMP3) significantly decreased in both plant species the values of the three tested parameters, suggesting that a photoinhibition process due to damage of the photosynthetic apparatus may have occurred (Li et al., 2014) (Tables 4.5 and 4.6). The functional alteration of the photosynthetic apparatus in duckweed plants exposed to different toxic compounds has been previously studied by the analysis of chlorophyll

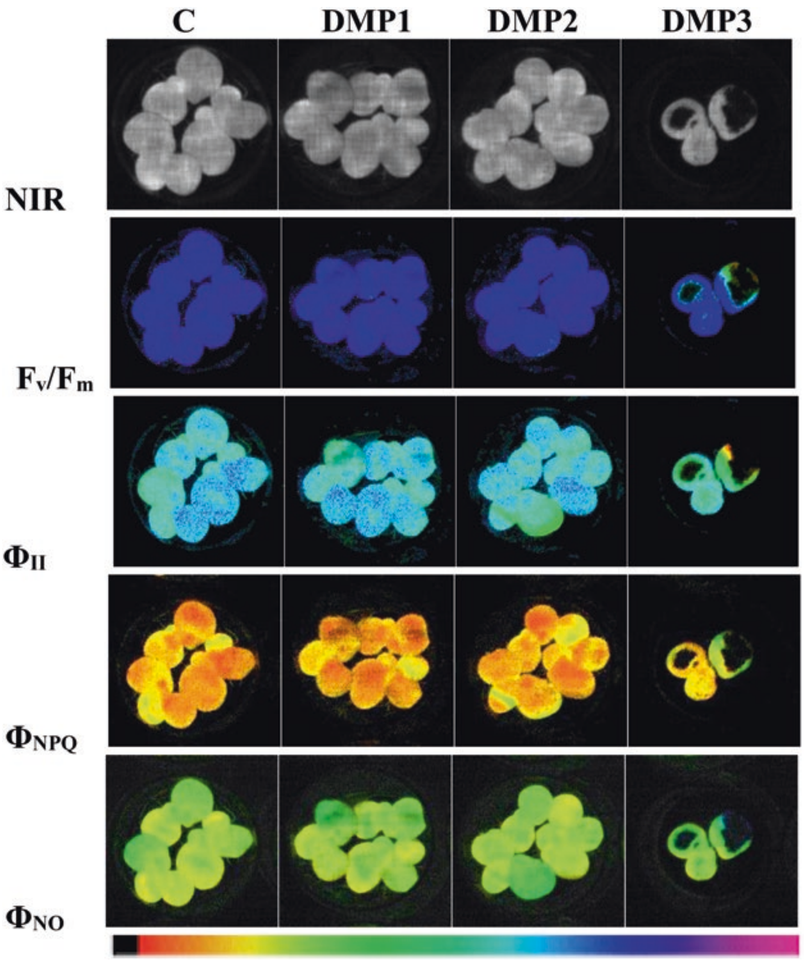

Fig. 4.2 Near infrared (NIR) and chlorophyll fluorescence images of maximum quantum yield of PSII photochemistry (F_v/F_m) in dark-adapted fronds and the effective PSII quantum yield (Φ_{II}) and the quantum yield of regulated (Φ_{NPQ}) and non-regulated (Φ_{NO}) energy dissipation in PSII at steady-state with actinic illumination of 60 μmol photons m^{-2} s^{-1} measured at the end of the experiment (7 days) in fronds of *Spirodela polyrhiza* (L.) Schleid. treated with different DMP concentrations (*C*, 0 mg/L (control), DMP1, 3 mg/L; DMP2, 30 mg/L; DMP3, 600 mg/L). The false colour code depicted at the bottom of the images ranges from 0.000 (black) to 1.000 (pink)

fluorescence parameters (Di Baccio et al., 2017; Pietrini et al., 2019; Alkimin et al., 2019; Hájková et al., 2019).

To further investigate the potential non-standard endpoints highlighting the effects of DMP on duckweeds at the physiological level, spectral reflectance properties of duckweed fronds were investigated by analysing some spectral indices obtained using an ASD FieldSpec-3 spectroradiometer. In this regard, as pointed out by Peñuelas and Filella (1998), it is to be underlined that leaf reflectance properties

Table 4.5 Light response curves of chlorophyll fluorescence parameters (photosynthetic efficiency (α), maximum electron transport rate (ETR_{max}), and the minimum saturating irradiance (E_k)) measured in *Lemna minor* L. fronds treated with different DMP concentrations for 7 days

Parameter	*C*	DMP1	DMP2	DMP3
α	0.318 ± 0.008 a	0.299 ± 0.008 a	0.295 ± 0.007 a	0.263 ± 0.016 b
ETR_{max}	32.1 ± 1.9 a	34.1 ± 3.4 a	33.2 ± 3.1 a	14.5 ± 1.3 b
E_k	101.8 ± 8.4 a	117.8 ± 13.7 a	112.1 ± 9.6 a	56.3 ± 5.3 b

C (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

Table 4.6 Light response curves of chlorophyll fluorescence parameters (photosynthetic efficiency (α), maximum electron transport rate (ETR_{max}), and the minimum saturating irradiance (E_k)) measured in *Spirodela polyrhiza* (L.) Schleid. fronds treated with different DMP concentrations for 7 days

Parameter	*C*	DMP1	DMP2	DMP3
α	0.290 ± 0.003 a	0.278 ± 0.011 a	0.282 ± 0.012 a	0.230 ± 0.007 b
ETR_{max}	29.5 ± 1.4 a	27.4 ± 1.1 a	28.9 ± 0.8 a	12.2 ± 0.8 b
E_k	101.8 ± 4.5 a	99.7 ± 6.1 a	103.5 ± 3.8 a	52.7 ± 2.5 b

C (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

are due to leaf surface and internal structure characteristics, besides concentration and distribution of biochemical molecules. For this reason, leaf reflectance spectra represent a valuable tool to monitor the physiological status in plants under different stress conditions (Croft & Chen, 2017). The values of six spectral reflectance indices, measured in our case study at the end of the 7-day treatment in *Lemna* and *Spirodela* fronds, are respectively reported in Tables 4.7 and 4.8 (for parameter description see Sect. 4.2). At a general outlook, in both plant species, it can be observed that the exposure to DMP1 and DMP2 did not cause substantial variations of most parameters, whereas in DMP3 a significant alteration in the values of all the spectral indices was clearly appreciable. Then, a substantial agreement of these parameters with both standard endpoint (chlorophyll content) and non-standard ones (chlorophyll fluorescence parameters) can be highlighted. In particular, the photochemical reflectance index (PRI), a proxy for physiological variables of light-dependent photosynthetic reactions (e.g. effective PSII quantum yield (Φ_{II}) or NPQ) that is increasingly used also as an index of photosynthetic performance (Kováč et al., 2018; Porcar-Castell et al., 2012), showed a similar trend to that described for the effective PSII quantum yield (Φ_{II}) in both plant species (Tables 4.7 and 4.8). Moreover, this index put in evidence that the damaging effect of the highest DMP concentration was more pronounced in *Spirodela* than in *Lemna* plants. This trend

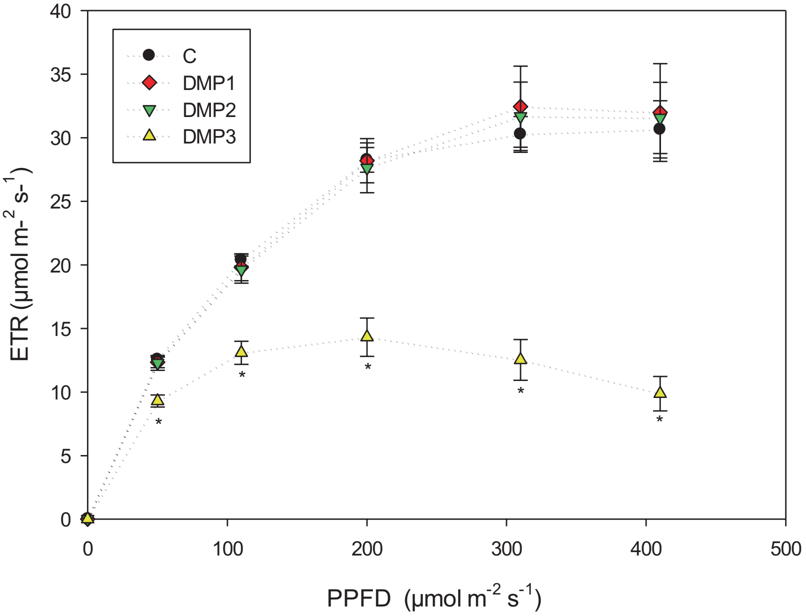

Fig. 4.3 Light response curves of the electron transport rate (ETR) in *Lemna minor* L. fronds treated with different DMP concentrations for 7 days: *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data points and vertical bars represent means ($n = 8$) ± S.E., respectively. At each light intensity, significant differences in ETR values compared to control are indicated with asterisks ($^*P < 0.05$)

was also confirmed by the analysis of the other indices related to chlorophyll content, namely, pigment specific simple ratio a (PSSRa), pigment specific simple ratio b (PSSRb), and normalised phaeophytinization index (NPQI), respectively associated with the chlorophyll a and b content and to the level of chlorophyll degradation (Blackburn, 1998; Peñuelas et al., 1995; Peñuelas & Filella, 1998). Thus, these non-standard parameters highlighted in both species a similar pattern to that observed for a standard one like the chlorophyll content (Tables 4.3 and 4.4). In fact, particularly at the highest DMP concentration, the reduction of the values of spectral parameters associated with chlorophyll pigments (PSSRa + PSSRb), with respect to control, was quite similar to the reduction of chlorophyll content observed in Tables 4.3 and 4.4. At the same time, normalised phaeophytinization index (NPQI), that is, a typical stress indicator of phaeophytinization, i.e. degradation of phaeophytin, a pigment involved into charge separation in PSII (Peñuelas et al., 1995), exhibited a quite similar reduction percentage compared to control (~60%) in both species when exposed to DMP3 (Tables 4.7 and 4.8). On the contrary, the carotenoid content, measured using the pigment specific simple ratio c (PSSRc) (Blackburn, 1998), showed a different trend reducing its values, with respect to

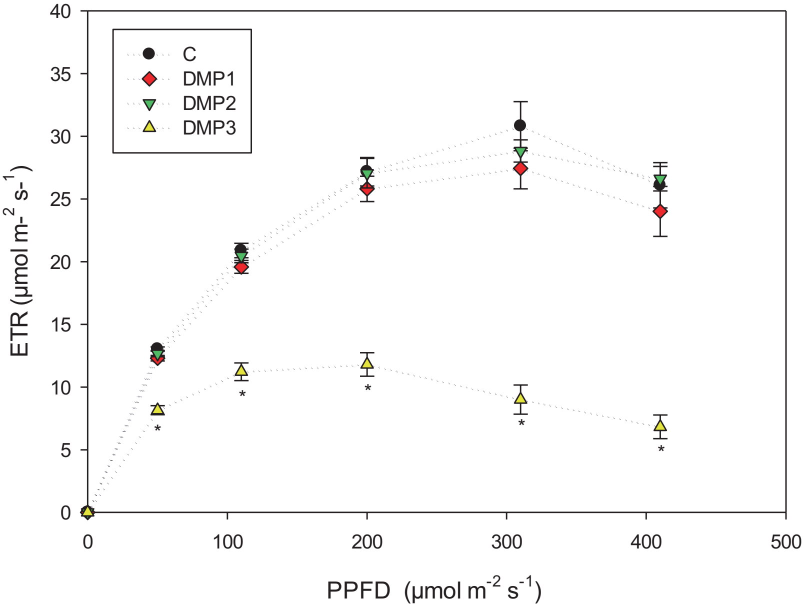

Fig. 4.4 Light response curves of the electron transport rate (ETR) in *Spirodela polyrhiza* (L.) Schleid. fronds treated with different DMP concentrations for 7 days: *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data points and vertical bars represent means ($n = 8$) ± S.E., respectively. At each light intensity, significant differences in ETR values compared to control are indicated with asterisks ($*P < 0.05$)

control, of 60% and only 18%, in *Spirodela* and *Lemna* plants, respectively, exposed to DMP3. As reported in the literature, carotenoids play essential roles in photosynthesis and photoprotection (Havaux et al., 2000; Sun et al. 2018b). In this study, a slight reduction of carotenoid content in *Lemna* plants exposed to DMP3 did not lower the ability of plants to cope with the stressing situation caused by DMP by enhancing energy dissipation. Finally, the anthocyanin reflectance index (ARI) values, associated with the anthocyanin content (Gitelson et al., 2001, 2009), increased after the treatment of duckweed plants by DMP, not only in both species under DMP3 condition but also in *Lemna* plants treated with the lower DMP concentrations (DMP1 and DMP2). This allows arguing about a stress-related defence response commonly ascribed to the photoprotective role of anthocyanin in plants (Pietrini et al., 2002; Gould, 2004). The analysis of the spectral reflectance indices confirmed the higher ability of *Lemna* plants to maintain the photoprotective strategies that alleviate photoinhibition and are associated with avoiding light absorption and successfully consuming or dissipating the light energy absorbed by light-harvesting pigments (Giossi et al., 2020). As already underlined, despite the

Table 4.7 Values of spectral indices in *Lemna minor* L. fronds treated with different DMP concentrations for 7 days

Parameter	*C*	DMP1	DMP2	DMP3
PRI	0.031 ± 0.002 a	0.034 ± 0.001 a	0.036 ± 0.002 a	0.010 ± 0.003 b
$PSSR_a$	7.213 ± 0.634 a	7.438 ± 0.508 a	7.335 ± 0.450 a	5.075 ± 0.375 b
$PSSR_b$	3.756 ± 0.186 a	3.810 ± 0.162 a	3.768 ± 0.156 a	2.953 ± 0.145 b
$PSSR_c$	7.118 ± 0.691 a	7.631 ± 0.583 a	7.372 ± 0.592 a	5.871 ± 0.422 b
NPQI	0.037 ± 0.006 a	0.040 ± 0.005 a	0.049 ± 0.012 a	0.015 ± 0.004 b
ARI	0.091 ± 0.025 c	0.200 ± 0.059 a	0.200 ± 0.050 a	0.158 ± 0.027 b

Treatments are *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

PRI photochemical reflectance index, $PSSR_a$ pigment specific simple ratio for Chl_a, $PSSR_b$ pigment specific simple ratio for Chl_b, $PSSR_c$ pigment specific simple ratio for carotenoids, *NPQI* normalised phaeophytinization index, *ARI* anthocyanin reflectance index (ARI)

Table 4.8 Values of spectral indices in *Spirodela polyrhiza* L. fronds treated with different DMP concentrations for 7 days

Parameter	*C*	DMP1	DMP2	DMP3
PRI	0.041 ± 0.002 a	0.037 ± 0.002 a	0.039 ± 0.001 a	−0.005 ± 0.006 b
$PSSR_a$	17.07 ± 2.60 a	13.69 ± 2.51 a	13.11 ± 1.63 a	6.681 ± 0.989 b
$PSSR_b$	7.330 ± 0.602 a	6.630 ± 0.712 a	6.414 ± 0.232 a	3.334 ± 0.431 b
$PSSR_c$	18.96 ± 3.44 a	14.52 ± 2.84 a	14.00 ± 2.12 a	7.77 ± 1.06 b
NPQI	0.093 ± 0.020 a	0.066 ± 0.019 ab	0.053 ± 0.014 ab	0.030 ± 0.005 b
ARI	−0.030 ± 0.007 b	−0.039 ± 0.005 b	−0.118 ± 0.035 c	0.118 ± 0.045 a

Treatments are *C* (plants grown in Hoagland's nutrient solution without DMP); DMP1 (plants grown in Hoagland's nutrient solution with 3 mg/L DMP); DMP2 (plants grown in Hoagland's nutrient solution with 30 mg/L DMP); and DMP3 (plants grown in Hoagland's nutrient solution with 600 mg/L DMP). Data are the mean values of 8 replicates ± standard error (SE). One-way ANOVA was applied, and data followed by different letters in the same row are significantly different (Tukey's test, $P \leq 0.05$)

PRI photochemical reflectance index, $PSSR_a$ pigment specific simple ratio for Chl_a, $PSSR_b$ pigment specific simple ratio for Chl_b, $PSSR_c$ pigment specific simple ratio for carotenoids, *NPQI* normalised phaeophytinization index, *ARI* anthocyanin reflectance index (ARI)

relevant information obtained on plant physiological status, leaf reflectance spectra evaluation in duckweed plants exposed to stress conditions is not a common feature. In fact, to our knowledge, only two works dealing with this issue are reported by using *L. gibba* (Dong et al., 2002) and *L. minor* (Iriel et al., 2015) plants, while more literature is present for other aquatic plants (Iriel et al., 2015; Song & Park, 2020).

4.4 Conclusion

In the case study presented in this work, two different sets of information were obtained. First, the whole data set acquired by analysing the responses of two duckweed species, *L. minor* and *S. polyrhiza*, to different DMP concentrations confirmed the damaging role exerted by this phthalate compound at 600 mg/L, both at biometric and physiological levels. Then, remarkable indications about non-standard toxicity endpoints were obtained by applying for the first time the spectroscopic techniques for the acquisition of chlorophyll fluorescence images and leaf reflectance spectra to evaluate the effect of a phthalate compound, namely DMP, in plants. In fact, by paralleling the responses of the two plant species to the three different DMP concentrations in terms of standard endpoints (i.e. biometric and chlorophyll content) and non-standard ones (i.e. leaf chlorophyll fluorescence parameters and reflectance spectral indices), interesting evaluations about the suitability of physiological proxies to assess the toxicity status of plants could be made. In particular, among parameters commonly adopted to describe the status of the photosynthetic machinery, a set of indices were highlighted as highly fitting with the information obtained by other more commonly used endpoints. By using the chlorophyll fluorescence technique, for instance, it was possible to put in evidence that parameters analysed in this study were capable to effectively describe the damaging effects exerted by DMP at the physiological level, paralleling those observed with standard endpoints as biometric indices and chlorophyll content. Further, parameters associated with light response curves (LCs) of electron transport rate (ETR) also emerged as valuable indicators of the toxicity status in aquatic plant leaves exposed to a toxic compound. Finally, through the acquisition of leaf reflectance spectra, the indices reported in this research were recognised as suitable to highlight the adverse effects exerted by a xenobiotic on duckweed photo-physiology.

Therefore, it can be claimed that leaf chlorophyll fluorescence parameters and reflectance spectral indices can be used as valuable non-standard proxies to investigate the toxicity effects of chemicals by using information-rich, non-destructive, and real-time mode technologies to assess the photo-physiological status of aquatic plants.

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Chapter 5
Pesticides: Impacts on Agriculture Productivity, Environment, and Management Strategies

Bilal Ahmad Khan, Muhammad Athar Nadeem, Humaira Nawaz, Muhammad Mohsin Amin, Ghulam Hassan Abbasi, Muhammad Nadeem, Muhammad Ali, Muhammad Ameen, Muhammad Mansoor Javaid, Rizwan Maqbool, Muhammad Ikram, and Muhammad Ashar Ayub

Abstract Pesticides are noxious organic and inorganic compounds used to kill or restrict the population expansion of harmful organisms. Pesticides have been used for a long period to kill pests and protect crops. Pesticides have been in use since the

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B. A. Khan · M. A. Nadeem · M. M. Amin · M. M. Javaid
Department of Agronomy, College of Agriculture, University of Sargodha, Sargodha, Punjab, Pakistan

H. Nawaz
College of Earth and Environmental Sciences, University of the Punjab, Lahore, Punjab, Pakistan

G. H. Abbasi · M. Ali · M. A. Ayub (✉)
Institute of Agro-Industry and Environment, The Islamia University of Bahawalpur, Bahawalpur, Punjab, Pakistan

M. Nadeem
Institute of Agro-Industry and Environment, The Islamia University of Bahawalpur, Bahawalpur, Punjab, Pakistan

Institute of Soil and Environmental Sciences, University of Agriculture, Faisalabad, Pakistan

M. Ameen
Institute of Agro-Industry and Environment, The Islamia University of Bahawalpur, Bahawalpur, Punjab, Pakistan

Department of Soil Sciences, Faculty of Agriculture and Environment, The Islamia University of Bahawalpur,
Punjab, Pakistan

R. Maqbool
Department of Agronomy, University of Agriculture, Faisalabad, Punjab, Pakistan

M. Ikram
Department of Botany, PMAS, Arid Agriculture, University Rawalpindi, Rawalpindi, Pakistan

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early 1940s, when dichlorodiphenyltrichloroethane (DDT) was first launched, ushering in a new era in man's struggle against pests and pathogens. Pesticide technologies have continued to generate a wide range of pesticides, providing adequate food supply to meet consumer demand, and these pesticides are considered an important tool for crop protection and growth, but they are harmful to the environment. Pesticide overuse can lead to biodiversity loss and destruction. Biodiversity is critical to human survival on our planet. Pesticides are harmful to a variety of creatures, including birds, aquatic animals, and mammals. Pesticides are a key source of anxiety for the long-term survival of our planet. This chapter will discuss the pesticide groups, their use, and impact on the environment. This chapter also discusses pesticide contamination and the long-term consequences of pesticides on the entire ecosystem. An alternative pest management and control strategy, such as integrated pest management (IPM), can be helpful to minimize the number and volume of pesticide treatments by integrating several control measures, like cultural control, the use of resistant genotypes, and physico-mechanical control along with chemical control. Furthermore, advances in biotechnology and nanotechnology may make it simpler to develop herbicides with fewer adverse effects or resistant genotypes along with a lower dose of herbicides.

Keywords Organic and inorganic compounds · Crop protection · Environment · Physico-mechanical control · Pesticides · Herbicides

5.1 Introduction

The demand for food will increase with the increase of the world's population to over 10 billion people by 2050 (Bahar et al., 2020). To meet this demand, increased food production is the most important goal of the globe. Global food production has increased by 70%, according to the UN's Food and Agricultural Organization (Saravi & Shokrzadeh, 2011). To meet the rising food demand, herbicides, insecticides, fungicides, and nematicides are frequently used to lower the pest pressure (Sarkar et al., 2021). Organic and inorganic pesticides and fertilizers are now widely used in agriculture production (Baweja et al., 2020). In 1940, synthetic insecticides like organochlorine (OCl) were used for the first time to control pest populations (İkizoğlu, 2020). Before using these synthetic pesticides, non-chemical control measures such as physical, mechanical, and cultural methods were used to control the pest (Folorunso et al., 2021). Pesticides have become a necessary part of modern agriculture to protect the crop from pest insects or disease organisms. Pesticides cost around $38 billion per year around the world for this purpose (Germany, 2012). To meet global demand, researchers and manufacturers are developing new formulas for pesticides. Most pesticides, however, are not ideal since they are toxic to other creatures, bioaccumulate, and are not ecologically friendly (Nadeem et al., 2020a). According to studies, approximately 0.1% of pesticides kill just target species, while the rest contaminates the ecosystem (Carriger et al., 2006). Pesticides are harmful to the environment's air, water, and soil. Pesticides have also been able to become a part of the food chain due to bioaccumulation. Pesticide exposure can cause both acute and chronic health problems for humans, animals, and aquatic life (Mostafalou et al., 2012).

5.2 History of Pesticide Use

Pesticides were initially employed in Ancient Rome to kill pests by burning sulfur. They also used slats, bitters, and ashes for weed control. A Roman suggested the use of arsenic as a pesticide (Mahmood et al., 2016). In 1600, a mixture of arsenic and honey was used to manage the ant population. In the late 1800s, farmers in the United States used nicotine sulfate, sulfur, and calcium arsenate to kill the pests of crops, but the results were unsatisfactory, owing to their crude application method (Delaplane et al., 2000). In 1867, copper and arsenic were used in impure form to manage a Colorado potato beetle infestation in the United States (Mahmood et al., 2016). Pesticides used before 1940 are considered first-generation pesticides and include inorganic (mineral) pesticides such as lead, mercury, and copper. These have a lengthy half-life and can accumulate in fatty tissues. Organic insecticides, often known as botanicals, are derived mostly from plants and break down rapidly in the environment. Synthetic botanicals are second-generation pesticides, also called post-1940s pesticides. This product was made by tampering with natural botanicals. Inorganic pesticides include arsenic, copper, lead, and mercury salts. Some of these herbicides are quite hazardous. These retained for a long time in the environment and are typically non-degradable. These are neurotoxins that induce tremors and developmental delays in children. Pesticides became more frequently utilized after World War II when effective pesticides could be made at a low cost. Pesticides such as dichlorodiphenyltrichloroethane (DDT), aldrin, dieldrin, benzene hexachloride, chlordane 2,4-dichlorophenoxyacetic acid, and endrin were introduced in 1939 (Delaplane et al., 2000). Between 1950 and 1955, fungicides (captan andglyodin) and organophosphate insecticide (Malathion) were introduced. Triazine herbicide group was discovered between 1955 and 1960 (Mahmood et al., 2016). From 1961 to 1971, an experimental military pesticide, recognized as Agent Orange, used in Vietnam War was introduced (Olson & Cihacek, 2020).

5.3 Classification of Pesticides

Pesticides are considered to be important to crop protection and disease treatment in humans. Pesticides are classified as repellents, destroyers, or targets of organisms. Insecticides sold in stores are ineffective against a variety of insects and pests (Speck-Planche et al., 2012). Commercial pesticides are also used in large quantities to control pests. Pests are treated prophylactically and remedially depending

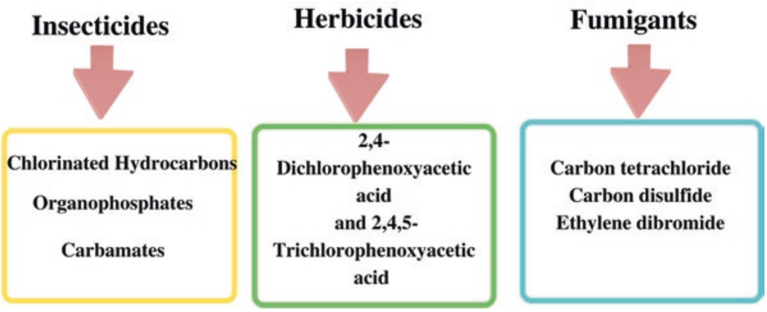

Fig. 5.1 Major classification of pesticides

upon pest species and control measures. Pesticides classified by their target species include insecticides, herbicides, fungicides, molluscicides, nematicides, plant growth regulators, and rodenticides. Pesticides used in massive quantities have detrimental effects on the environment. According to research, pesticides are affecting badly human health and biodiversity (Agrawal & Sharma, 2010). Pesticides are typically water-soluble and stable, and they affect natural flora, fauna, and aquatic life. In agricultural applications, pesticides cause toxicity, as well as occupational exposure to workers in the manufacturing process (Rashid et al., 2010) (Fig. 5.1).

Pesticides come in a variety of shapes and sizes, each targeted to a specific insect.

- Algaecides are chemicals that are used to slow down algae growth.
- Antimicrobials are used to keep germs such as viruses and bacteria under control to the greatest extent possible.
- Pesticides that are derived from living organisms are known as biopesticides.
- Disinfectants are chemicals that kill viruses, bacteria, as well as other germs.
- Foggers are used to kill insects that have come into contact with chemicals.
- Pesticides that are used to control fungal diseases are known as fungicides.
- Insecticides are the pesticides that are used to control insect pests.
- Miticides are chemicals that are used to reduce mite's species that feed on animals and plants. Insects are not technically mites.
- Slugs, snails, and other mollusks are controlled with molluscicides.
- Herbicides are pesticides that are used to suppress or eliminate weeds.

5.4 Impact of Pesticide Use in Agriculture

5.4.1 Target Effects of Pesticide on Organism

Pesticide development and application has expanded in recent years to aim at a large number of pests, but the frequency and increased pesticide quantity use has made it difficult for pests to adapt to new environments (Meyers & Bull, 2002; Cothran et al., 2013). Changes in gene mutation, population growth rate, and generation number helped the pests to adapt to their new environment.

5.4.2 *Resistance in Pests to Pesticides*

Resistance is described as a heritable modification in population of pest sensitivity. "Resistance can be defined as a heritable shift in the sensitivity of a pest population, as indicated by a product's repeated failure to achieve the expected level of control when used in accordance with the label guidelines for that pest species" (IRAC, 2014). Pesticide use has resulted in the resistance development in a number of targeted pest species all over the world (Tabashnik et al., 2009). Main crop pests, animal parasites, disease vectors, and common urban pests have all evolved resistance to the point that treatment is extremely difficult (Skendžić et al., 2021). Pesticide resistance is influenced by a number of elements, including genetics, biology/ecology, and pesticide management activities (Gage & Schwartz-Lazaro, 2019). One of the most widely used methods for detecting resistance is insecticide bioassays, despite major drawbacks. Various novel approaches for finding pesticide resistance have been developed in the recent two decades, including a combination of insecticide bioassays and advanced biochemical and molecular techniques. These methods include enzyme assays, enzyme electrophoresis, and other processes.

5.4.3 *Pest Resurgence*

Pest revival is demonstrated as rapid comeback of the pest population in dangerous figures following the usage of a pesticide (Skendžić et al., 2021). The use of specific, broad-spectrum, and persistent pesticides which have the ability to remove important natural enemies is considered as the main reason of pest comeback (Zaller, 2020). However, the resurgence has been documented for a lot of reasons, including in feeding of insect pest and propagation rates which is due to the use doses of sub lethal pesticide and the removal of a primary pest can sometimes create suitable environment for secondary pests to become major pest (Cusumano et al., 2020). Natural enemies such as ladybird beetles, spiders, mirid bugs, and various illnesses keep natural BPH populations under control. Pesticides, on the other hand, have affected BPH female fecundity, favoring their recovery (Table 5.1).

5.5 Non-target Effects of Pesticide on Organism

Pesticide effects on non-target organisms have been a cause of widespread interest and worry for decades. Pesticides' detrimental impacts on arthropods (non-target) have been thoroughly documented (Mehdizadeh et al., 2021). Unfortunately, natural insect foes such as parasitoids and predators are very vulnerable to insecticides and suffer severe repercussions (Sánchez-Bayo, 2021). Because natural enemies play an important role in pest population management, their disappearance might

Table 5.1 Effect of pesticides on plants

Name of pesticide (brand name)	Chemical formula	Active ingredient	Application dose per acre	Crop	Effect	Reference
Gramoxone	$C_{12}H_{14}Cl_2N_2$	Paraquat	1000–1250 ml	Sugarcane	All weeds	Aekrathok et al. (2021)
Leader	$C_{16}H_{18}N_6O_7S_2$	Sulfosulfuron	13.5 g	Wheat	All weeds	Dheer et al. (2021)
Regent	$C_{12}H_4Cl_2F_6N_4OS$	Fipronil	480 ml	Maize/ sugarcane	Termite, stem borer and top borer	Triques et al. (2021)
Topsin-M	$C_{12}H_{14}N_4O_4S_2$	Thiophanate methyl	100–150 g	Orchards	Powdery mildew and anthracnose	Avenot et al. (2020)
Dual Gold	$C_{15}H_{22}ClNO_2$	S-metolachlor	800 ml	Cotton/maize	All weeds	Kashe et al. (2020)
Tycon	$C_{56}H_{81}NO_{15}$	Emamectin benzoate	75 g	Cotton/ vegetables	Spotted and pink bollworms, jassid, and aphid	Mokbel and Huesien (2020)
Imidacloprid	$C_9H_{10}ClN_5O_2$	Imidacloprid	250 ml	Cotton	Jassid, aphid, whitefly, and thrips	Wumuerhan et al. (2020)
Puma S	$C_{18}H_{16}ClNO_5$	Fenoxaprop-p-ethyl	250–300 ml	Wheat	Wild oat	Yaman and Nalbantoğlu (2020)
Rifit	$C_{17}H_{26}ClNO_2$	Pretilachlor	400–500 ml	Rice	Broad leaved + sedges	Yaghoubi et al. (2020)
Bifenthrin	$C_{23}H_{22}ClF_3O_2$	Bifenthrin	250–330 ml	Cotton/ vegetables	Spotted and pink bollworms	Arshad et al. (2019)
Metribuzin	$C_8H_{14}N_4OS$	Metribuzin	80 g	Wheat	All weeds	Bhoite et al. (2019)
Stomp	$C_{13}H_{19}N_3O_4$	Pendimethalin	1000 ml	Wheat	All weeds	Khajavi et al. (2019)
Stomp	$C_{13}H_{19}N_3O_4$	Pendimethalin	1000 ml	Cotton/maize	All weeds	Iqbal et al. (2019)
Sunstar	$C_{15}H_{18}N_4O_7S$	Ethoxysulfuron	80 g	Rice	Broad leaved + sedges	Tomar et al. (2019)
Vimix	$C_{10}H_{11}ClN_4$	Acetamiprid	125 g	Cotton/ vegetables	Whitefly, jassid, aphid, and thrips	Ullah et al. (2019)
Click	$C_8H_{14}ClN_5$	Atrazine	1000 ml	Maize	All weeds	Verma et al. (2018)
Treflan	$C_{13}H_{16}F_3N_3O_4$	Trifluralin	1250–1500 ml	Cotton	All weeds	Karasali et al. (2017)

increase pest problems. If natural enemies are not present, additional pesticide treatments are usually required to manage the target insect (Haddi et al., 2020).

The natural enemies that preserve lesser pests in check are occasionally injured, resulting in subsequent pest outbreaks. Uncontrolled pesticide use in agricultural systems has wrought havoc on the population of soil arthropods, in addition to natural adversaries (Krupke & Tooker, 2020). Invertebrates such as mites, nematodes, micro-arthropods, springtails, spiders, insects, earthworms, and other minute creatures make up the soil food web, which helps digest leaves, plant wastes, and manure (Temegne et al., 2021).

5.5.1 Earthworms

Earthworms account for more than 80% of terrestrial invertebrates which convert organic material into humus by improving soil fertility (Yasmin & D'Souza, 2010). The earthworms are influenced by a range of agricultural practices, the most significant of which is the indiscriminate use of pesticides (Pelosi et al., 2013). Pesticide usage may have a negative impact on earthworm populations. Earthworms are particularly toxic to carbamate insecticides, for example, and various organophosphates have been shown to have a deleterious influence on earthworm populations (Pelosi et al., 2013). Similarly, field research in South Africa discovered that both long-term and intermittent exposure to chlorpyrifos and azinphos-methyl had deleterious effects on earthworms (Reinecke & Reinecke, 2007). According to various scientific research (Yasmin & D'Souza, 2010), pesticides changes in population of alter earthworm and their cocoons hatching but it depends upon dosage applied.

In several of the following examples, pesticides were the major causes of predator population decline:

- *Collembola* species range, evenness, and richness as well as spider numbers were decreased in chlorpyriphos-applied plots in comparison to control plots in grassland pastures (Sánchez-Bayo, 2021).
- Studies on the impact of pesticides on soil arthropods were conducted near Everglades National Park in the United States in an agricultural region. When non-treated fields were compared to fields sprayed with herbicides and insecticides, researchers revealed that non-sprayed fields had a larger number of arthropods (including predators like coccinellids and spiders) than sprayed ones (Ngin et al., 2017).

5.5.2 Pollinators

Pollinators are fascinating organisms that help in pollination. Honey bees, fruit flies, bumblebees (*Bombus* spp.), birds, and certain beetles are among the pollinators that have been discovered such as honeyeaters, sunbirds, and hummingbirds, and their actions are impacted by ecological factors produced by pathogens, competition, illnesses, pesticides, predators, and habitat changes; pollinators can be employed as bioindicators of exosystemic processes in a number of ways (Njeri, 2020). Pesticide use, on the other hand, leads to direct losses of insect pollinators as well as indirect agricultural losses owing to insufficient pollinator numbers (Riggi et al., 2022).

5.6 Pesticide Pollution

5.6.1 Effects of Pesticides on Humans

Harmful effects of chemical pesticides on health of humans are due to persistence of pesticides in the environment and the capability to enter the food chain. Pesticide's chemicals can enter the food chain of humans through different sources like vegetables and fruits, polluted air, and contaminated water. Chronic and acute disorders are caused by pesticides, which are listed below:

5.6.1.1 Pesticides and Their Consequences

- Pesticides have a tendency to travel large distances through the soil, water, and air.
- Washing off into rivers and streams, causing death or other health problems in fish (such as bone deterioration).
- Bioaccumulation is the accumulation of a long-lasting pesticide in the body (usually fatty tissues).
- Biological magnification when the biological accumulation in each organism is compounded.
- Pesticide concentrations are higher in those higher on the food chain (Table 5.2 and Fig. 5.2).

5.6.2 Environmental Effect of Pesticides

Pesticides have a tendency to travel large distances through the soil, water, and air. They can wash off into rivers and streams, causing death or other health problems in fish (such as bone deterioration). Acute disease usually develops within a few days of exposure to chemical. The drift of pesticides from agricultural areas, its exposure, and deliberate use should all be considered. Pesticides can cause a variety of symptoms, like skin rashes, body aches, headaches, dizziness, nausea, impaired eyesight, cramping, and in severe cases, coma and death (Germany, 2012). Quantity and toxicity of agents used, method of action, administration method, and frequency and length of contact with pesticides are all factors that influence the severity of these concerns (Richter & Schläpfer, 2002). Among all over the world, approximately three million cases of acute poisoning per year are recorded.

Table 5.2 Effects of pesticides on human health

Name of pesticide (brand name)	Chemical formula	Active ingredient	Effect on human	References
Topsin-M	$C_{12}H_{14}N_4O_4S_2$	Thiophanate methyl	Liver and thyroid effect	Moo-Muñoz et al. (2021)
Bifenthrin	$C_{23}H_{22}ClF_3O_2$	Bifenthrin	Abdominal pain, sore throat	Park et al. (2021)
Treflan	$C_{13}H_{16}F_3N_3O_4$	Trifluralin	Throat irritation, cancer	Rani et al. (2021)
Click	$C_8H_{14}ClN_5$	Atrazine	Tumor, breast, and uterine cancer, lymphoma	Rohr (2021)
Regent	$C_{12}H_4Cl_2F_6N_4OS$	Fipronil	Sweating, headache, nausea	Song et al. (2021)
Tycon	$C_{56}H_{81}NO_{15}$	Emamectin benzoate	Central nervous system depression, aspiration pneumonia	Niu et al. (2020)
Metribuzin	$C_8H_{14}N_4OS$	Metribuzin	Liver failure, organ disorder	Samir et al. (2020)
Rifit	$C_{17}H_{26}ClNO_2$	Pretilachlor	Gill cohesion, cell necrosis	Shilpakar et al. (2020)
Vimix	$C_{10}H_{11}ClN_4$	Acetamiprid	Nausea, respiratory failure	Ma et al. (2019)
Puma S	$C_{18}H_{16}ClNO_5$	Fenoxaprop-p-ethyl	Epigastric pain, vomiting	Tandon (2019)
Stomp	$C_{13}H_{19}N_3O_4$	Pendimethalin	Headache, nausea	Ansari et al. (2018)
Imidacloprid	$C_9H_{10}ClN_5O_2$	Imidacloprid	Tachycardia, nausea, vomiting	Mahajan et al. (2018))
Dual Gold	$C_{15}H_{22}ClNO_2$	S-metolachlor	Anemia, diarrhea	Quintaneiro et al. (2018)
Gramoxone	$C_{12}H_{14}Cl_2N_2$	Paraquat	Kidney failure, coma	Thi Hue et al. (2018)
Leader	$C_{16}H_{18}N_6O_7S_2$	Sulfosulfuron	Fast heartbeat, lung scarring	Sharma and Singhvi (2017)
Sunstar	$C_{15}H_{18}N_4O_7S$	Ethoxysulfuron	Nausea, headache	Sharma and Singhvi (2017)

5.6.3 *Illness That Lasts a Long Time*

Humans develop chronic sickness after being exposed to sublethal levels of pesticides for a long time (Germany, 2012). Chronic symptoms do not appear immediately and appear later. Agricultural laborers are more likely to be afflicted. However, the general public is affected, primarily as a result of tainted food and water, as well as chemicals floating from the fields (Germany, 2012). As pesticides have become a

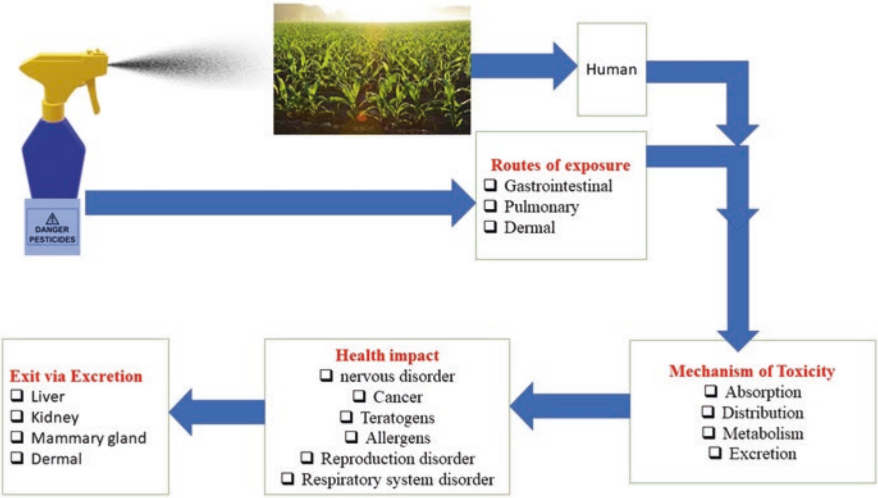

Fig. 5.2 Pesticide route of exposure, mechanism of toxicity, health impacts, and route excretion

larger part of our ecology, the incidence of chronic diseases has begun to rise. Pesticide exposure is linked to the occurrence of human chronic diseases disturbing the neurological, respiratory, reproductive, and cardiovascular systems, according to collected data (Mostafalou et al., 2012).

5.7 Pesticide Exposure

Human contact to pesticides may arise through industrial contact in the case of agricultural workers in greenhouses and open fields, pesticide industry workers, and executioners of house pests (Damalas & Eleftherohorinos, 2011). However, irrespective of whether the profession includes pesticide use, existence of such chemicals in the work environment constitutes a potential professional exposure. Clearly, workers who mix, load, transport, and apply manufactured pesticides are generally contemplated to be the group that will gain the most exposure due to work nature. And because of this, the potential for serious intoxication is high (Chen et al., 2001a) In some cases, pesticides may be exposed due to accidental chemical spill, leakage, or damaged spraying equipment; when workers do not pay proper attention on given instructions, exposure of workers increases; it happens mostly when they ignore the basic safety measures and basic sanitation like handwashing after the use of chemicals.

Some factors can disturb exposure during pesticide handling (Chen et al., 2001a).

- Pesticide formulation may affect the exposure extent. There is a risk of splashing and occasional spray of liquids, resulting in direct or indirect contact with the

skin through clothing contamination. Solid substances can create dust when loaded into application devices, resulting in exposure of the face and eyes, as well as respiratory threats.
- Pesticide product packaging can also affect exposure. For example, pesticide bag openings can affect the exposure depending on the combination with formulation of active ingredient.
- Furthermore, adjuvant chemicals used in pesticide formulations to increase their efficacy in relation to biological activity may show harm themselves, thus contributing to overall exposure effect to a product (Chen et al., 2001b).

5.8 Pesticides and the Loss of Biodiversity

Several pesticides are harmful to beneficial insects, mammals, birds, or fish. The wildlife poisoning depends on the toxicity of pesticides, applied quantity, timing, frequency, application method, weather, and soil type. Over the past four decades, the use of organophosphate and carbamate has strongly increased. In the south, organochlorines (endosulfan) are highly persistent in the environment. With the habitat change, majority of the population has declined due to pesticide poisoning (Gibbs et al., 2009). Weed and insect population which are the food sources of many species has decreased due to pesticides. By altering the vegetation structure, herbicides can change the habitats ultimately leading to a decline in population (Boatman et al., 2004).

- Pesticides influence wildlife directly and indirectly via habitat and food sources.
- Wildlife poisoning by highly toxic herbicides, insecticides, fungicides, and rodenticides can cause major population decline.
- Pesticides accruing in the food chain, especially those which cause endocrinal disturbance, pose a long-term risk to birds, mammals, fish, and amphibians.
- Broad-spectrum herbicides and insecticides decrease food sources for mammals and birds which ultimately leads to decline in infrequent species populations.
- Herbicides can reduce habitats unsuitable for certain species by changing vegetation structure. This threatens mammals, farmland birds, and insects.

5.9 Pesticide Impact on Soil Environment

Pesticides which are being used for agriculture and other purposes accumulate in soil in considerable amounts. Pesticides are used extensively and repeatedly, increasing the problem of soil formation. Fate of applied pesticides is determined by a variety of parameters such as characteristics of soil and soil microflora, and it endures numerous degradation, adsorption, desorption, and distribution processes (Hussain et al., 2010). The pesticides that have deteriorated cooperate with the soil, soil microbes, and biochemical processes, affecting microbial diversity and enzyme activity (Hussain et al., 2010). Pesticide impacts and their various mechanisms in soil are as follows:

1. Pesticides which get into the soil can change soil biomass and soil microbial diversity. Any variation in actions of soil microorganisms as a result of pesticide application causes soil ecosystem disruption and soil fertility loss (Mishra et al., 2016). Numerous studies have shown the negative effects of these on soil microbes and soil respiration (Sofo et al., 2012). Exogenous pesticide treatments may also affect the purpose of useful root-colonizing microbes, such as bacteria, algae, and fungi, in soil by disturbing their growth, metabolic activity, and other factors. These can get into soil and cooperate with soil microorganisms in a variety of ways: It can have a negative impact on the growth, variety, and soil microflora biomass. Sulfonylurea herbicides such as chlorsulfuron, thifensulfuron-methyl, and metsulfuron-methyl have been shown to exhibit growth of Pseudomonas bacteria (Ahemad & Khan, 2011a). *Pseudomonas* spp. are recognized to have a significant ecological role in soil habitat (Ahemad & Khan, 2011a, b), and so their eradication can have a negative impact on soil fertility. In the same way, captan, benomyl, and chlorothalonil have been shown to reduce respiration of soil by 30–50% (Chen et al., 2001a, b)
    (a) Pesticides can also inhibit or kill specific microorganisms, allowing them to outnumber and outcompete other groups (Hussain et al., 2010). Endosulfan application resulted in a 76% increase in bacterial biomass while reducing 47% fungal biomass (Xie et al., 2011).
    (b) Some microbial groups may use pesticides as energy source, increasing their growth and causing changes in soil ecology. Bacterial detaches from wastewater-irrigated agricultural soil demonstrated capability to use chlorpyriphos (carbon source) for their growth (Bhagobaty & Malik, 2010).
    (c) Pesticides can change and reduce the functional structure and diversity of microorganisms resulting in reduction in microbial activities in soil and biomass (Lupwayi et al., 2009).
2. Pesticides can disrupt critical biochemical events in the soil, such as nitrification, ammonification, and nitrogen fixation, by deactivating or activating particular soil microbes and enzymes (Hussain et al., 2010). Rate of soil biochemical responses is finally governed by the harmonious and additive relations among pesticides, microorganisms, as well as soil characteristics (Srinivasulu et al., 2012).
3. These have been shown to affect the soil organic mineralization, a main soil feature that influences soil quality and production. An important decrease in soil organic matter was discovered after applying herbicides (glyphosate, atrazine, paraquat, and primeextra) (Sebiomo et al., 2011). However, after continued

application from the second to sixth week of herbicide treatment, soil organic matter increased.

4. Pesticides that get into the soil can interrupt local breakdown or change the enzymatic activity of the soil (Floch et al., 2011). The soil has an enzymatic pool that includes free enzymes and enzymes secreted by microorganisms, all of which are indications of biological balance, such as soil quality and fertility (Hussain et al., 2010).

5.10 Weed Flora and Pesticides

Understanding the connections among weed control measures and weed range, as well as properly managing cultivation system to avoid weed appearance and maintain seed (weeds) bank in the soil low, is critical for successful weed management (Chen et al., 2001a). All farming systems require proper cultivation and farming practices for the management of weeds, such as a well-organized crop rotation, early sowing dates, sowing of cover crops, and proper row spacing (Andersen et al., 2012). In contrast, chemical weed management should be based on a thorough understanding of the environmental conditions that influence herbicide effectiveness both before and after application (Chen et al., 2001b).

5.11 Pesticide Management Strategies

Some pesticide resistance management strategies have been planned that are free from risk and successful in many situations, for example, monitoring pest populations in the field before applying pesticides, altering pesticides with different modes of operation, limiting many applications over space and time, generating refuges, and avoiding gratuitous persistence. The most difficult part of managing resistance is to ensure that it is effectively adopted by producers and pest control operators, not their shortcomings (Dhaliwal et al., 2016).

Pesticides must be used more efficiently in the current situation to decrease environmental pollution while boosting their efficiency against target pests. Pesticide resistance and pest comeback can be reduced in this way. As a result, consideration has been given to their uses and their physical and environmental selection. Differential toxicity among species for a given pesticide characterizes physiological selectivity. Ecological selectivity, on the other hand, refers to the alteration of operating processes to prevent wasteful destruction of non-target organisms (Gill & Garg, 2014). The farmers should focus on the insecticides employed that are more poisonous to board species than their natural opponents, which may help to limit revival (Dhaliwal et al., 2016).

5.11.1 Integrated Pest Management (IPM)

Farmers' Field School, Farmers' Training Program, Method and Result Demonstration, PlantWise Program, and print and electronic media have all collaborated with the integrated pest management (IPM) program to educate farmers. The pest warning, on the other hand, was begun by a separate wing in order to send out alarms.

The IPM program regulates pesticide use by marking the quantity of sprays for certain pests, such as weed, insect, and plant diseases, which cause crop losses while also putting human health at risk. However, the country confronts difficulties in terms of pesticide use and management. As a result, the major issues with pesticide management are related to incorrect pesticide selection, high rates of application, and uncounted sprays.

In addition, to provide effective crop protection, extension personnel and growers require knowledge of climate change, the emergence of new pests, and newly registered chemicals. It is vital to have this knowledge and then to apply pesticides according to the instructions in order for the Department of Agriculture Extension efforts and money invested on pesticides to reflect delivering the potential use to growers. With the rising concern for environment and human health, we must guarantee that the authorities' recommendations are followed, which includes avoiding the indiscriminate use of pesticides to protect applicators, food consumers, and environmental and human health. This pamphlet has been updated to include chemicals registered in Pakistan, as well as target pests and crops for pesticide selection and application.

When it comes to pest treatment, consider using an integrated pest management method, as these approaches are meant to cause the least amount of environmental disruption. IPM strives to reduce pesticide indiscriminate use and replace dangerous chemicals with safer alternatives. IPM is a method of accomplishing long-term, ecologically friendly pest control via technology use and other pest management techniques. IPM refers to an ecological approach in pest management in which all available necessary techniques are consolidated in a unified program so that populations can be managed in such a way that economic damage is avoided and adverse side effects are minimized. For creating sustainable maize-based cropping systems in European cultivable systems, farmer incentives and multidisciplinary research are important to uplift the use of new IPM strategies. These IPM approaches can donate to Europe's planned assurance to pesticide use that is environmentally sustainable (Vasileiadis et al., 2011). Growers may be hesitant to use an IPM strategy because of the additional expense and effort involved, but the government and extension services can assist to persuade and encourage farmers to use IPM plans to control environmentally benign and long-lasting pest control. We've previously addressed how long-term pesticide use leads to pesticide resistance and a rebound of pests. We can continuously go for further possible management solutions, such as physical and cultural control, biocontrol, biopesticides, and so on, to prevent these concerns.

5.11.2 Cultural Management

Historically, farmer's most significant defense to prevent crop damage was cultural management measures. As it is cost-effective and ecologically natural, farmers around the world have long used cultural control for pest management (Gill et al., 2013). Regular farm procedures designed to remove pests from causing plant harm are known as cultural control practices. Cultural management strategies include resistant cultivar, rotation of crops, soil solarization, allelopathy, intercropping, organic mulches, and farmyard manure (Dhaliwal et al., 2016). Soil solarization and organic mulches alone and their combination have been reported as an economical and environment-friendly technique for reducing arthropods and weeds on surface of soil. By merging these strategies with life cycle of pest, more active cultural control can be achieved. Farmers eliminate large insect populations by revealing them to severe climatic problems through agricultural methods such as hoeing, weeding, and plowing. Plowing agricultural fields provides for soil turnover in the upper layer during weed burying and eliminating wastes from the previous year. In grain maize (*Zea mays* L.) and sorghum (*Sorghum bicolor* L.) farms in South Africa, for example, slicing the plants killed roughly 70% of hibernating spotted stalk borer. Discing and plowing of plant remain after reducing killed a further 24% of grain sorghum population and 19% of maize population. In India, planting dates, barrier crops, and pearl millet were considered to be beneficial against maize stem borer (Sultana et al., 2012). Rodents are considered as pests in India's agriculture, forestry, and animal husbandry industries, as well as in people's homes, horticulture, and urban and rural storage facilities. Adequate tillage of soil, clean cultivation, crop scheduling, barriers, proofing, and repellents are examples of cultural measures that can limit rodent harborage, food supplies, and immigration (Begum et al., 2017).

5.11.3 Physical and Mechanical Control

Physical control is controlling the population of insects with the help of devices that affect them physically. Steaming woolen winter clothing, for example, aids in the eradication of woolly bear moth (Dhaliwal et al., 2016). Many concealed pests, such as bulb bugs and eelworms, are killed by treating plant storage items like bulbs and corns with hot water (Dhaliwal et al., 2016). Mechanical control denotes to manual devices used to control insect populations. Handpicking, trapping devices, cutting, and pruning are some of the techniques used. The most profitable and extensively used starling control method is said to be trapping using live lure birds (Campbell et al., 2012). Several wildlife species, including squirrels, coyotes, and birds, have been identified as pests of California agriculture. Various non-lethal strategies, including exclusionary devices, baiting, and habitat modification, are often favored for these pests (Baldwin et al., 2013). Management of weeds in growing crops by using mechanical method is mostly related with tillage operations using particular instruments such as hoes, brushes, and harrows. Increased awareness of herbicide side effects has fueled interest in mechanical weed management, resulting in an increase in the predominance of organic practices (Jat et al., 2011). Tephritid flies can be controlled by trapping them with yellow-colored sticky traps (Dhaliwal et al., 2016).

5.11.4 Use of Nanotechnology

Pesticides produced in nanomaterials for agricultural uses, whether particularly static on a hybrid substrate or functionalized nanocarriers for external stimuli or enzyme-mediated triggers, are referred to as nano-pesticides. Nanosized particles are expected to investigate pesticide actions in nanocarrier novel formulations based on a variety of materials, including lipids, polymers, copolymers, silica, ceramics, metals, and others (Andrea et al., 2000).

Nano-pesticide formulations can improve water solubility and preserve agrochemicals from degradation in the environment, revolutionizing disease, weed, and insect control in crops (Abdullah et al., 2011). The cytotoxicity and genotoxicity of nanomaterials, on the other hand, are on the edge.

Pesticide use that is indiscriminate and irrational disrupts the ecosystem's balance and puts everyone's health at risk. Short-term and long-term adverse effects of accidental absorption of pesticide residues from food and water consumption are lethal or result in disability. Children are more sensitive to pesticides and are more likely to suffer irreversible organ and tissue damage as a result. The central and peripherical neurotoxicity and the effect on blood coagulation capacity are all valid reasons for concern (Arinze & Yubedee, 2000). Indeed, a thorough examination of the advantages and disadvantages that affect the activity and toxicity of nano-pesticides is critical for safe and long-term growth of nanoparticles in agriculture, which has already been approved.

The effect of compositions on nano-pesticide behavior in the atmosphere, ecologies, agricultural workers, customers, and all the other associated industries in the agriculture chain is unknown (Armstrong et al., 2013). Nano formulations, from the other hand, are well renowned for their critical role in minimizing active ingredient degradation, promote water emulsification equilibrium, and ramping up active substance biological availability, particularly to reduce agricultural goods and types of food to prevent endemic mosquitoes, plant injury, and economic damage (Armstrong et al., 2013).

Nano pesticides from agricultural and manufacturing sewage treatment stormwater enter the groundwater via soil perforating dissolution after a rainfall event, altering its quality, advancing human sun exposure, and posing ecological concerns. Because of their biomimetic properties and high natural tendency of distribution and biomagnification in soil, surrounding waters, types of food, and, as a consequence, all animals, especially mammals, nanoparticles were found to have toxic effect repercussions (Agrawal & Sharma, 2010; Ahemad et al., 2009; Baldwin et al., 2013). For humans, the variety of side effects associated with particular vulnerability and exposure duration to nanomaterial results in acute and long-term pathological embodiments such as inhalation, cardiovascular risk, lymphatic diseases, immunodeficiency, cognitive impairment, and brain tumors that can express following the exposure or generations later than the rest of bioavailability (Band et al., 2011) and depending on unique nanoparticle properties (Arinze & Yubedee, 2000; Bhagobaty & Malik, 2010).

5.11.5 Allelochemicals Are Used to Control Pests

Weeds, insects, and diseases all pose a risk to crop plants, preventing them from reaching their genetic potential and limiting yield. Pesticides are used to manage agricultural pests in approximately 3.5 million tons worldwide. Herbicides account for nearly half of these, with insecticides (30%), fungicides (15%), and other chemicals accounting for the remaining 5%. Pesticides cause significant harm to the soil, air, and water. Pollution of the environment by "pesticides" is common in almost all agrarian civilizations, including Pakistan (Nadeem et al., 2020b, c).

Herbicide use in indefinite period increases the amount of herbicide biotypes while also polluting the environment, significantly reducing quality of the food, and negatively affecting animal health (Nadeem et al., 2021c; Maqbool et al., 2021b).

Allelopathy is an instinctual and ecologically responsible phenomenon characterized by the inhibition or top player effects of phytopathogens produced by agricultural crops, also recognized as phyto-chemicals (Nadeem et al., 2020b, 2021b; Maqbool et al., 2021a) whenever it minimizing pest insects, diseases, and bindweed in agricultural production, it holds great potential. Allelopathy, on the other side, has been shown to be extra effective in farmland integrated pest management.

Allelopathic harvests can aid in weed management in a variety of ways, including the use of potentially toxic agricultural residues as leaf litter and crop rotation.

• Allelopathic crop rotations • Allelopathic crop intercropping • Allelopathic crop water extracts.

Synthetic pesticides can be used in conjunction with biotic and abiotic drip irrigation extracts, and their efficacy may be increased when combined with synthetic pesticides. In crop moisture leaf extract from multiple crops, instinctual various chemicals such as metabolites, astringency, phenylpropanoids, phenolics, essential oils, representation, flavonoid, commonly used in project management, and quercetin have been unearthed (Nadeem et al., 2021a; Maqbool et al., 2021c).

It is safer for such environment to use fewer herbicides, and the harmful effects of artificial pesticides may be lowered. Pesticide, greater goods are required to compete in the global marketplace and procure deep discounts. Drip irrigation extracts are a somewhat expensive starting point of herbicide application than synthetic chemicals, which reduces farmers' manufacturing costs. Similarly, using this procedure can reduce the cost of trying to import weed killers (Rs 3 billion per year).

5.11.6 Chemical Management

Chemical restrictions (pesticides) seem to be the most commonly used method in many integrated management projects. Pest populations (invertebrates, harmful bacteria, raccoons, and so on) are reduced to thresholds which does not damage the crop using contaminant controls. Long before the advent of chemical pesticides after Second World War, cultivation oil is made, spur inorganic salts, soaps, lime sulphur, nicotine, and just few both these inorganic compounds such as sulphur and metals were used as herbicides. The modern pesticide era started with the discovery of highly effective pesticides such as dichlorodiphenyltrichloroethane (DDT) and organophosphates (OP).

Their remarkable effect against plant pathogens led to their misuse, actually resulting in a downturn that eventually led to the development of an alternative technique, IPM. Synthetic regulations were viewed as one tool in a significantly bigger strategy to reduce the harmful consequences of pests on crops by IPM. Due to their greater and often instantaneous influence on a pathogen population, chemical regulations should be used as a last luxury hotel in an IPM program.

5.12 When Should Pesticides Be Applied?

It is essential to consider the timeframe of any agrochemicals before implementing it. When used correctly, it can have the greatest impact on particular living things while having minimal repercussions on beneficial microorganisms. The accessibility of forecast window frames, the best time to resolve the problems, and the lowest percentage of harm to non-target lifeforms and the surroundings all influence herbicide implementation timing.

During agricultural blossoming and the intermediate of the day, mosquitoes are more vulnerable to pesticides. Monitoring insects in the field is thus necessary in order to determine the phase of parasitic insects. When implementing pesticides, the duration during the day and summer is also an important consideration (Gentz et al., 2010) for insect pests for which surveillance systems are still not obtainable; spray dictatorships or experimental studies must be carried out to ascertain the much more ideal moment for pyrethroid administering (Gentz et al., 2010).

Because high winds are more likely to happen in the mid afternoon as the surface temperature rises, the best moments for herbicides are typically early morning and late afternoon. As a consequence, thermal convection heat begins to rise conveniently and effectively mixes with cooler air above it, promoting drift. Pesticides that drift endanger important pollinator insects, consists of various, selectively bred pets, and natural areas. Sprinkling at air currents with speeds significantly larger than 2.5 miles per second is simply not recommended since it can lead to serious dispersion and damage to the environment in neighboring areas (Gentz et al., 2010). Pesticides should not be applied too close to storms because the rain can wash pesticides and herbicides away without endangering the intentional pest.

5.13 Pesticide Dosage

To achieve the best outcomes, the herbicide dose should really be sufficient but not disproportionate. The fungicide producer calculates the medication to achieve an appropriate amount of control, satisfactory residual oil levels, and economic resource per component of formulated insecticide. Continual pesticides require less anticipated needs than anti-pesticide residues, and they last longer on the target; however, when using continual pesticides and herbicides, use caution because at small concentrations, they can reduce the benefit availability of natural enemies. The probability of imidacloprid residues negatively impacting natural predators increases dramatically when an insect repellent is persistent (Gentz et al., 2010).

5.14 Pesticide Placement

Herbicides should be applied in the field in order to achieve maximum specific organization. Only about 1% of pesticides used achieve their designated destination, resulting in a significant amount of waste. Recognizing pest biology and behavior is critical because that can continue providing relevant data on the pest's ecosystems, fecundity, getting fed, and other considerations to take into account and then use herbicides. Because most herbicides are sprayed as fluids, the size of the droplets is critical for their effectiveness. Smaller droplets provide good insurance and have a greater chance of reaching the destination than larger droplets, which speedily rebound off the surface of the plant.

5.15 Concluding Remarks and Future Prospects

Pesticides had been originally used to improve human life and increase agricultural production and try to prevent bacterial infections, but their negative effects now exceed the advantages. The previous discussion shows the devastation caused by unrestrained pesticide application on different environmental constituents. To mention a few consequences of pesticide use, there has been an increase in conferring resistance pest and diseases, a loss of advantageous creatures including such predators, microbial diversity of soil changes, pollinators, earthworms and contamination of the water and air ecosystems.

Because of synthetic insecticides' lengthy nature, herbicides had already found their way into numerous food chains and trophic levels, including living beings as well as other larger animals. As a result of consumption polluted water, air, or food, a few really patients with chronic human diseases now have evolved. This is the time to use pesticides correctly in a manner that safeguards our environment and, inevitably, our health. Alternate solution of pest management techniques, including such Pest management, might reduce the number and quantity of herbicide treatment options by utilizing various methods of control including based on culture control, use of isolated strains, physico-mechanical regulate, and reasoned herbicide use.

Furthermore, advances in biotechnology and nanotechnology might very well make it much easier to develop herbicides with lesser side effects or conferring resistance genotypes. Community participation and various extension programs that can start educating and encouraging farm owners to embrace innovative IPM practices are critical to reducing the negative impacts of pesticide residues on our surroundings. In the long term, synthetic pesticides may be used in combination with therapeutic activities and remedial measures and actually result in lengthier nuisance and insect control.

This combined effect guarantees ecological responsibility which can be used in a wide assortment of methods to manipulate urban insects and introduced species (Gentz et al., 2010). Pesticides have also affected badly the biological veracity of surrounding aquatic ecosystems. To fully comprehend indirect pesticide effects on the environment, studies from different disciplines such as conservation biology, pharmacology, population biology, biological ecology, community ecology, and landscape ecology must be combined (Macneale et al., 2010).

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Chapter 6
Occurrence, Distribution, and Fate of Emerging Persistent Organic Pollutants (POPs) in the Environment

Lubna Najam and Tanveer Alam

Abstract Persistent organic pollutants (POPs) are a group of organic molecules that resist natural degradation, bioaccumulate in the food web and organisms, and have serious health consequences. POPs are vulnerable to long-range atmospheric transport (LRAT), resulting in transboundary deposition, due to their lengthy residence duration in the climate and semi-volatility nature. POPs are poisonous, resistant to degradation, and bioaccumulative and have a wide spatial distribution, all of which have been related to mutagenic, reproductive, and immunological diseases. A global accord was reached at the Stockholm Convention to reduce and eventually eradicate the discharge of POPs directly into the environment. Organohalogenated chemicals, polycyclic aromatic hydrocarbons (PAHs), and pesticides are examples of persistent organic pollutants (POPs) that can be produced in a wide range of environmental compartments. Their existence in the aquatic environment is a global issue, with sediments serving as a store and hence a source of hydrophobic, stubborn, and toxic substances. Furthermore, because these pollutants vary in their capacity to bioaccumulate in tissues, they may impact the reproduction and death of living species. Organochlorine insecticides, polychlorinated biphenyls, and dioxins are among the synthetic compounds classified as persistent organic pollutants. Environmental persistence, transboundary movement, lipophilicity, and bioaccumulation are all characteristics of these organic pollutants, which might result in biomagnification-induced death, especially in top-ranked biota. POPs accumulate in the oceanic ecosystems, which is why they are so important. Despite the ubiquity of POPs and the biotic impacts they induce, there is a lack of information on their occurrences and their destiny in the environment. Organohalogenated chemicals, polycyclic aromatic hydrocarbons, and pesticides are examples of persistent organic pollutants (POPs) that can be found in a variety of environmental compartments. Their existence in the aquatic environment is a global issue, with sediments serving

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L. Najam (✉)
DAV(PG) College Muzaffarnagar, CCS University, Meerut, India

T. Alam
Natural & Medical Sciences Research Center (NMSRC), University of Nizwa, Nizwa, Oman

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as a storehouse and, as a result, a source of hydrophobic, persistent, and toxic substances. The presence of these persistent hazardous compounds in sediments indicates that aquatic contamination has occurred as a result of agricultural, industrial, and urban discharges, raising concerns about possible dangers to aquatic species, wildlife, and humans. This chapter aims to provide an updated comprehensive overview of the occurrence, distribution, and fate aspects of POPs. In addition, the chemical structures and properties of specific POPs are discussed.

Keywords Persistent organic pollutants · Biota · Organohalogenated compounds · Pesticides · Polycyclic aromatic hydrocarbons · Sediments · Environment · Bioaccumulation

Abbreviations

ABL	Atmospheric boundary layer
AMAP	Arctic Monitoring and Assessment Programme
BFRs	brominated flame retardants
BMR	benchmark response
CACAR	Canadian Arctic Contaminants Assessment Report
DDA	2,2-bis (p-chlorophenyl) acetic acid
DDD	1,1-dichloro-2,2-bis (4-chlorophenyl) ethane
DDE	1,1-dichloro-2,2-bis (4-chlorophenyl) ethylene
DDT	1,1,1-trichloro-2,2-bis (4-chlorophenyl) ethane
HBCD	hexabromocyclododecane
HCB	hexachlorobenzene
HCHs	hexachlorocyclohexane
HL	half-life
IPCC	Intergovernmental Panel on Climate Change
Kow	octanol/water partition coefficient
LRAT	long-range atmospheric transport
LRET	long-range environmental transport
LRT	long-range transport
NE	North-East
OCPs	organochlorine pesticides
OCs	organochlorine compounds
PAH(s)	polycyclic aromatic hydrocarbon(s)
PBB	polybrominated biphenyls
PBDDs	polybrominated dibenzo-p-dioxins
PBDE	polybrominated diphenyl ethers
PBDFs	polybrominated dibenzofurans
PBTs	bioaccumulative toxic chemicals
PCB	polychlorinated biphenyls
PCDBF-2,3,4,7,8	pentachlorodibenzofuran
PCDDs	polychlorinated dibenzo-p-dioxins
PCDFs	polychlorinated dibenzofurans

PFA	perfluoroalkoxy
PFOS	perfluorooctanesulfonic acid
POPs	persistent organic pollutants
QWAI	quantitative water air sediment interaction
TBBP-A	tetrabromobisphenol A
TCDD-2,3,7,8	tetrachlorodibenzo-p-dioxin
TCDF-2,3,7,8	tetrachlorodibenzofuran
UNEP	United Nations Environment Program

6.1 Introduction

6.1.1 Persistent Organic Pollutants (POPs)

Over the last few years, there has been growing concern around the world about the health effects of environmental pollution on the population. Persistent organic pollutants are one kind of pollution that is emitted into the environment and are getting a lot of attention these days because of the dangers they pose.

Persistent organic pollutants (POPs) are a class of organic molecules of natural or synthetic origin that resist natural degradation through chemical, biological, and photolytic processes (UNEP, 1999). POPs are transported from one place to another place via air-water. As a result, POPs have spread across the globe, including to places where they have never been used before (Jones & Voogt, 1999).

POPs are persistent and they experience biomagnifications and bioaccumulation. POPs have a high potential for long-distance transport due to their persistence, enabling them to travel from the mid-latitudes to the NE Arctic regions. Despite major reductions in production or prohibitions of numerous persistent organic pollutants, POPs have continued to contaminate the climate and concentrate in food chains in recent years. They may also be harmful to public health and the environment. These chemicals pose a threat to the entire global community since they can be carried to locations where they are never produced. Under specific ambient temperatures, POPs can enter the gaseous phase and volatilize into the atmosphere from soils, vegetation, and water bodies.

Polycyclic aromatic hydrocarbons (PAHs) and pesticides are some names of persistent organic pollutants which may be produced in a wide range of environmental compartments. Their existence in the aquatic environment is a global issue, with sediments serving as a store and hence a source of hydrophobic, stubborn, and toxic substances. PAH concentrations in some lake sediments are expected to be high enough to impact sediment-dwelling creatures (Sanders et al., 1996; Rippey, 1990).

The large number of substances that may be found in tissue samples from living organisms from all over the globe (Muir, 1994; Ross et al., 2000) offers a significant challenge to policymakers charged with regulating the industry and conserving the environment.

POPs are commonly referred to as “poisons without passports.”

6.1.2 Common Properties of Persistent Organic Pollutants (POPs)

Persistent organic pollutants (POPs), overall, have several characteristics:

POPs are persistent in the environment: Persistent organic pollutants are a class of organic molecules of natural or synthetic origin that resist natural degradation through chemical, biological, and photolytic processes.

POPs are semi-volatile ($Pv < 1000$ Pa): Long-range atmospheric transport (LRAT) of POPs can lead to transboundary deposits, due to their semi-volatile properties and lengthy residence duration in the environment.

POPs have high bioaccumulation and biomagnification potential: They are found in higher concentrations near the top of the food chain.

POPs have low water solubility: POPs are poorly soluble in water but highly soluble in fats and oils. Persistent chemicals with these qualities bioaccumulate in living beings' fatty tissues easily.

Toxic to humans and wildlife: Even at the very low amounts present in the environment, wildlife, and humans, POPs have the potential to harm humans and other living things.

6.1.3 Classification of Persistent Organic Pollutants (POPs)

If an organic compound fits the following conditions, it is categorized as a POP.

1. To be persistent and have a long half-life in air, water, land, or sediment. For compounds in soil and sediment, the half-life (HL) must be greater than 180 days, 60 days for substances in water, and more than 2 days for pollutants in the air.
2. Capacity for long-distance transportation through air, water, or migratory species (Wania & Mackay, 1996).
3. Properties of bioaccumulation and bioamplification (Deribe et al., 2011; Hong et al., 2015; Wania & Mackay, 1996).
4. Toxic, hurts human health as well as the environment (Ashraf et al., 2015).

Persistent organic pollutants are categorized based on their uses and sources (Fig. 6.1). POPs can be of natural origin like dioxins and dibenzofurans originated from volcanic activity and vegetation fires (Jacob, 2013) or of anthropogenic origin – chemicals used in agriculture, such as pesticides (aldrin, chlordane, dieldrin, DDT, endrin, heptachlor, toxaphene, mirex, hexachlorobenzene), and in the industry (PCBs, HCB, BMR compounds) (El-Shahawi et al., 2010; Ferreira, 2013).

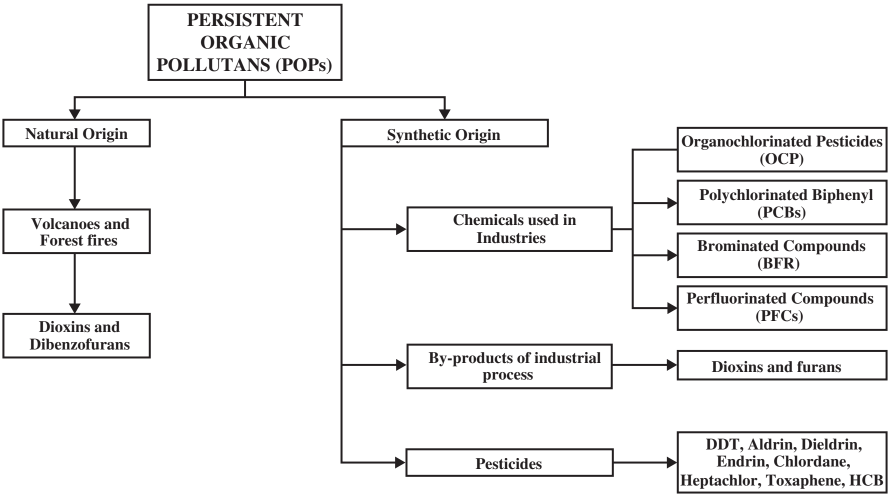


Fig. 6.1 Classification of persistent organic pollutants (POPs)

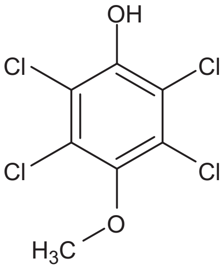


Fig. 6.2 Chemical structure of Drosophilin A

Natural Origin

Drosophilin A (p-methoxytetrachlorophenol) is an organic molecule that closely resembles POPs. Industrial persistent organic pollutants are halogenated compounds that are directly linked to environmental harm.

Drosophilin A (Fig. 6.2), initially identified in 1952 from a basidiomycete, a fungus whose spores form in basidia, was the first natural chlorinated chemical investigated and characterized to include a halogenated benzene ring (Anchel, 1952).

6.1.4 Types of Persistent Organic Pollutants (POPs)

During the post-World War II industrial revolution, many POPs were widely used. Although many of these compounds proved to be effective in the management of pests and diseases, many had unintended consequences for human health and the

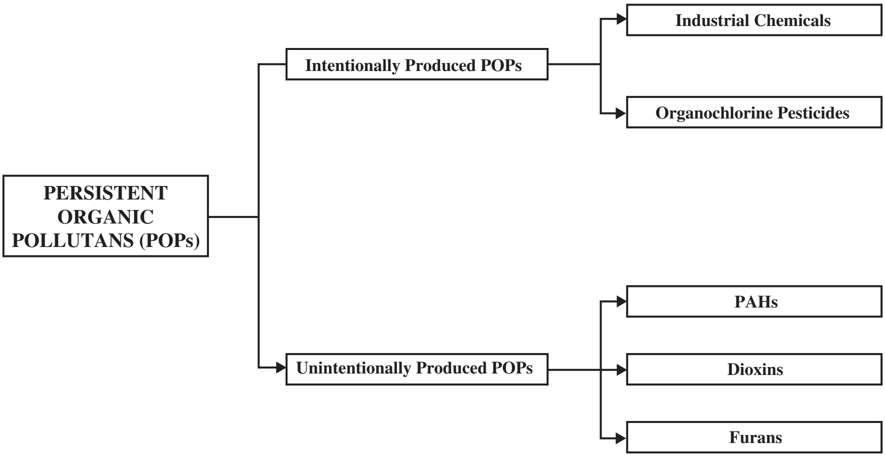


Fig. 6.3 Types of POPs – intentional and unintentional POPs

environment. Furthermore, POPs can be classified into two types: intentional and unintentional POPs as shown in Fig. 6.3.

1. Intentionally produced chemicals that are currently or have previously been utilized in agriculture, pest control, or industrial operations.

   For example, in industrial applications, PCBs are used, and dichlorodiphenyltrichloroethane, in many regions of the world, is used to control mosquitoes.
2. Unintentionally produced chemicals, like dioxins and dibenzofurans, which are produced by the decomposition of medical waste, volcanic activities, forest fire, and some chemical processes in the industries.

6.1.4.1 Intentionally Persistent Organic Pollutants (POPs)

Different chemical processes involving chlorine will create these chemicals as desired products. Such types of organic molecules are having chlorine atoms inside their structures. These compounds have high lipophilicity and high neurotoxic activity and are known as organochlorine compounds (OCs), e.g., DDT (dichlorodiphenyltrichloroethane), DDA, and PCBs (polychlorinated biphenyls) (Harner et al., 2006; Kelce et al., 1995).

Organochlorine Pesticides (OCPs)

Organic pesticides having chlorine in their composition are known as organochlorine pesticides (OCPs). A large number of organochlorine compounds are used around the globe. Chlorinated organic chemicals are manufactured in large quantities for industrial, agricultural, medicinal, and domestic uses. Generally, organochlorine insecticides are having neurotoxicity in living organisms. Because of the bioaccumulation process, they can be found throughout the food chain. (Stoytcheva, 2011; Jokanović, 2012). Aldrin, dieldrin, heptachlor, and endrin are common examples of OCPs. Despite their early promise, organochlorine pesticides are currently employed considerably less frequently due to their environmental pollution effects (Strompl & Thiele, 1997; Aspelin, 1994).

Pesticides containing organochlorines are divided into three categories:

1. *Dichlorodiphenylethanes:* p,p'-dichlorodiphenyltrichloroethane, p,p'-dichlorodiphenyldichloroethane, p,p'-dichlorodiphenyldichloroethylene, methoxychlor, perthane, etc.
2. *Chlorinated cyclodienes:* Hexachloro-hexahydro-endo-exo-dimethanonaphthalene (aldrin), hexachloro-octahydro-epoxy-dimehanonaphthalene (dieldrin), hexachloro-oxapentacyclotridecene (endrin), heptachloro-tetrahydro-methanoindene (heptachlor), hexachloro-hexahydro-methano-benzodioxathiepine-3-oxide (endosulfan), etc.
3. *Hexachlorocyclohexanes:* Benzene hexachloride (BHC), dodecachlorooctahydrometheno-cyclobuta-pentalene (mirex), octachlorocamphene (toxaphene), etc (Fig. 6.4).

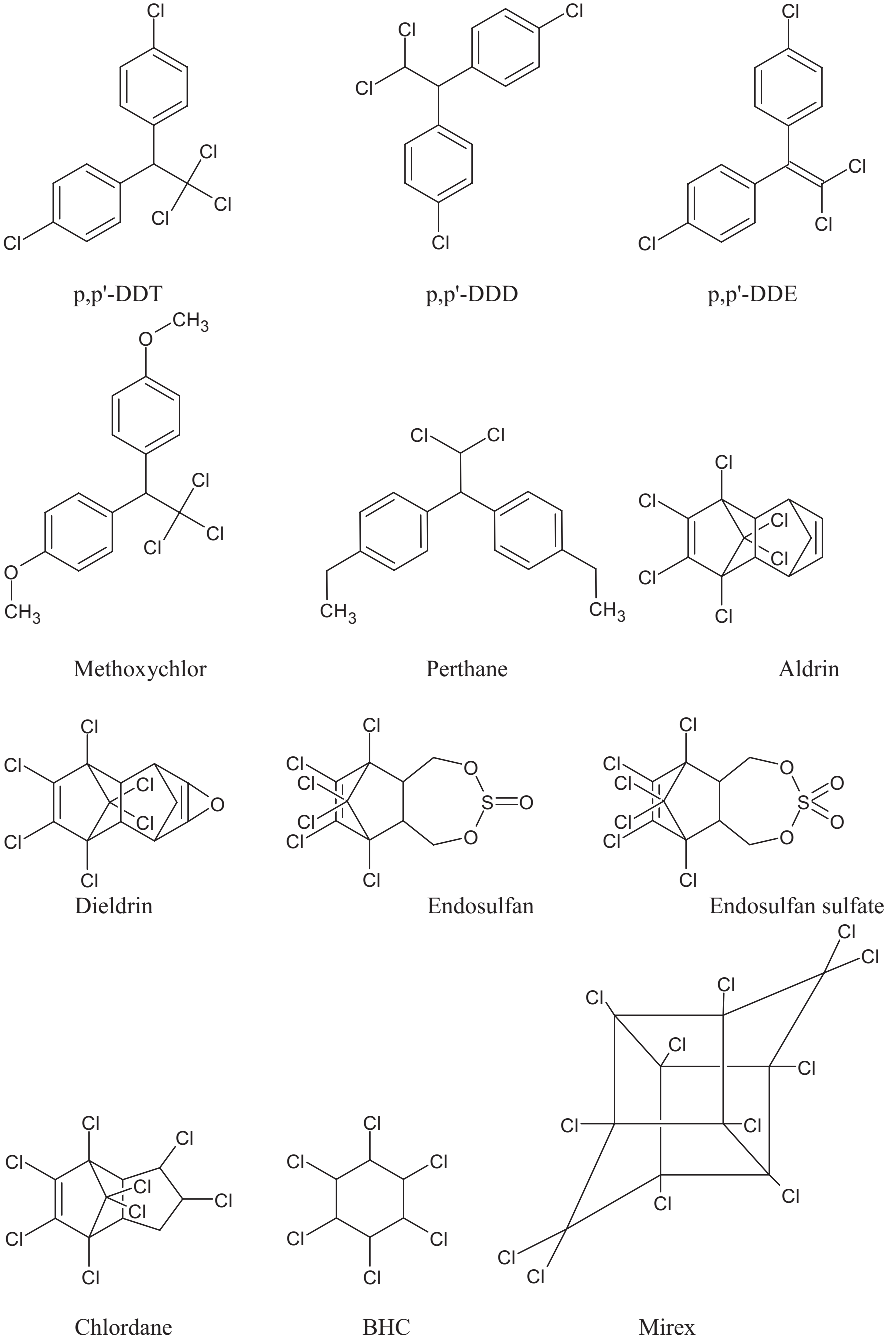
Industrial Chemicals

Polychlorinated Biphenyls (PCBs)

PCB compounds were manufactured by the chemical reaction of biphenyls with chlorine in the presence of various catalysts. Depending on the chlorine atom's number and their locations in the molecule, two hundred nine possible congeners are present in the environment (Fig. 6.5). Many PCBs cannot be degraded and allowed to remain for a longer duration in the climate and distributed through gaseous and water systems of transportation (Villa et al., 2003; Panic & Gorecki, 2006; Korytar et al., 2002). They are known as PCB dioxin-like congeners, and they are classified as endocrine disruptors and human carcinogens (Erickson, 2001; Zuçato et al., 1999).

Brominated Compounds

Organohalogens are a wide class of substances that include brominated flame retardants (BFRs). They are mostly used as fire retardants, as the names indicate. These chemicals are found in a wide range of consumer goods also. Hexabromocyclododecane (HBCD), tetrabromobisphenol A (TBBP-A), polybrominated biphenyls (PBB), polybrominated dibenzofurans (PBDFs), and polybromodiphenyl ethers are the most commonly used brominated flame retardants (Fig. 6.6). Consumption of contaminated food, ingestion, and inhalation of dust are the main causes of BFRs and exposure pathways.



p,p'-DDT p,p'-DDD p,p'-DDE

Methoxychlor Perthane Aldrin

Dieldrin Endosulfan Endosulfan sulfate

Chlordane BHC Mirex

Fig. 6.4 Chemical structures of organochlorine pesticides (OCPs)

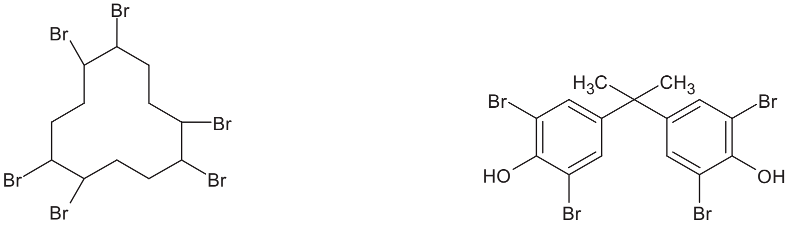
Polychlorobiphenyl (PCB52)

Polychlorobiphenyl (PCB80)

Polychlorobiphenyl (PCB126)

Polychlorobiphenyl (PCB99)

Fig. 6.5 Chemical structures of polychlorinated biphenyls



Hexabromocyclododecane (HBCD)

Tetrabromobisphenol A (TBBP-A)

Polybrominated biphenyl (PBB)

Polybrominated diphenyl ether (PBDE)

Fig. 6.6 Chemical structures of brominated compounds

6.1.4.2 Unintentionally Persistent Organic Pollutants (POPs)

They are undesired secondary products that are produced by the combustion or chemical processes of chlorine compounds. PAHs, dioxins, and furan compounds are the three categories of unintentional POPs.

Polycyclic Aromatic Hydrocarbons (PAHs)

Two or more fused benzene rings are organized in linear, angular, or cluster configurations in PAHs. These compounds contain only -C and –H atoms in their structures (Fig. 6.7). Stable carbon-carbon bonds hold the central molecular structure together. PAHs may exist in the climate naturally, primarily as a result of plant synthesis or following forest and prairie fires. The distribution of PAHs in sediments can reveal information on precursor sources, such as pyrogenic or petrogenic sources (Aboul-Kassim & Simueit, 1995; ATSDR, 1995; Zielinska & Samy, 2006).

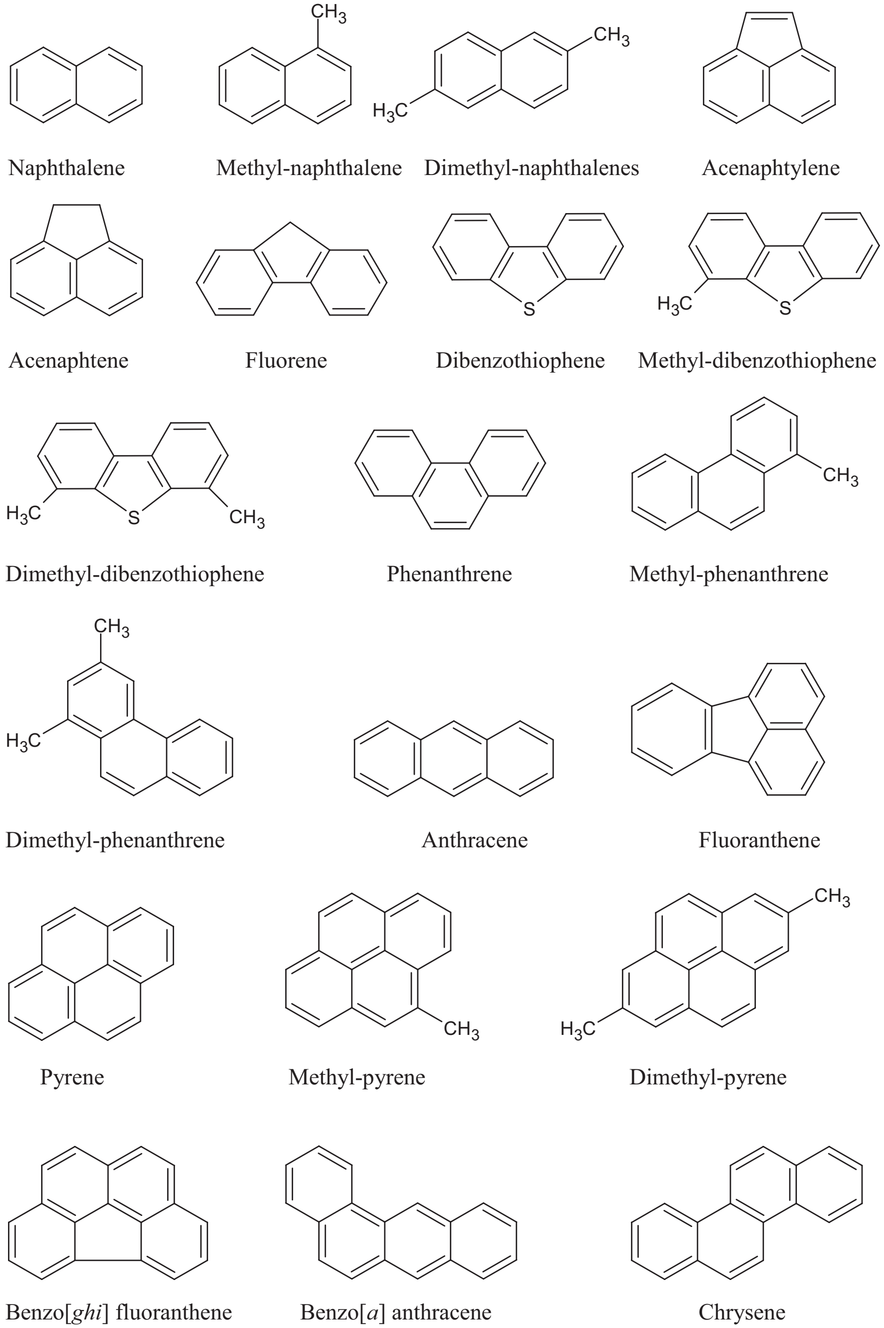
Dioxins and Furans

PCDDs (polychlorinated dibenzo-*p*-dioxins) and PCDFs (polychlorinated dibenzofurans) are extremely hazardous, and the most hazardous is TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin) (Van den Berg et al., 1998; Papadopoulos et al., 2004; Focant et al., 2004). Dioxins can form secondary products at higher temperatures, such as from urban sewage or various industrial operations. Dioxins are not used very often. Because of their high toxicity, PCDD/F can cause major issues in humans and other species even at low levels (US Environmental Protection Agency Health Assessment Document, 1994; Hirai et al., 2004). The chemical structures of TCDD, TCDF, and PCB are shown in Fig. 6.8.

6.1.5 Occurrence of Persistent Organic Pollutants (POPs)

POPs such as dioxins and furans are released into the climate in natural ways such as volcanic eruptions and forest fires. POPs are long-lasting and disintegrate slowly in the environment, as their title indicates. Drosophilin A was the first natural chlorinated chemical studied and characterized to have a halogenated benzene ring. It was discovered in 1952 in a basidiomycete, a fungus whose spores grow in basidia. Basidiomycetes are the ancestors of all well-known mushrooms and related species (Woitas, 2014). Table 6.1 shows the fewest number of species known to contain Drosophilin A.

Some of the most common anthropogenic man-made sources of POPs include incinerating plants, power stations, agricultural applications, thermal stations, and evaporation from surface waters, land, or landfills. Chemical facilities, wastes from the use of obsolete oil or burned oil, cement production, lixiviation of landfills, apparatus maintenance and repair, wastewater sludge, incineration of municipal waste, toxic and clinical waste, animal corpse cremation, fly ash disposal, and fossil fuel and different combustions are unexpected man-made sources of POP (Ying et al., 2005; Wenzl et al., 2006). POPs are transported to oceans through rivers and streams, where they are deposited as sediments on the bottom of these bodies of water and subsequently released.



Naphthalene

Methyl-naphthalene

Dimethyl-naphthalenes

Acenaphtylene

Acenaphtene

Fluorene

Dibenzothiophene

Methyl-dibenzothiophene

Dimethyl-dibenzothiophene

Phenanthrene

Methyl-phenanthrene

Dimethyl-phenanthrene

Anthracene

Fluoranthene

Pyrene

Methyl-pyrene

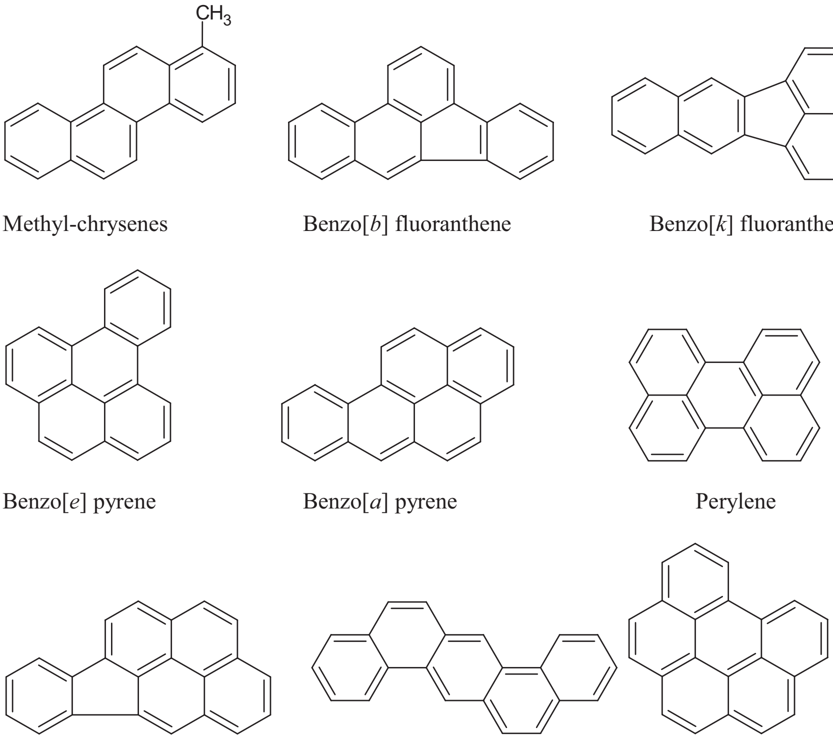
Dimethyl-pyrene

Benzo[*ghi*] fluoranthene

Benzo[*a*] anthracene

Chrysene

Fig. 6.7 Chemical structures of polycyclic aromatic hydrocarbons (PAHs)



Methyl-chrysenes Benzo[*b*] fluoranthene Benzo[*k*] fluoranthene

Benzo[*e*] pyrene Benzo[*a*] pyrene Perylene

Indeno[*1,2,3-cd*] pyrene Dibenzo[*a,h*] anthracene Benzo[*ghi*] perylene

Fig. 6.7 (continued)

2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) 2,3,4,7,8-pentachlorodibenzofuran (PCDBF)

Fig. 6.8 Chemical structures of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and 2,3,4,7,8-penta chlorodibenzofuran (PCDBF)

POPs can be discovered in both implemented atmospheres and in locations far from where they are employed. This is due to the application site's high stability, bioaccumulation ability, biomagnification potential, and potential to transit outside of the site of application. The breakdown of these compounds is extremely challenging. In the soils or sedimentary environments, the process of disintegration is much slower. Degradation mechanisms include photochemical degradation,

Table 6.1 Basidiomycetes mushrooms and related species containing Drosophilin A

S. No.	Name of organism	Quantity of Drosophilin A (mg/L)	Common name	References
1.	*Agaricus arvensis* Schaeff	0.500	Horse mushroom	Teunissen et al. (1997)
2.	*Bjerkandera adusta* (Willd.) P. Karst.	0.005	Smoky polypore or smoky bracket	Teunissen et al. (1997)
3.	*Coprinus plicatilis* Curtis) Redhead, Vilgalys & Hopple	20.0	Pleated inkcap	Bastian (1985)
4.	*Peniophora pseudopini*	1.00	–	Teunissen et al. (1997)
5.	*Phellinus fastuosus* (Lev.) Ryvarden	15.7	–	Teunissen et al. (1997), Singh and Rangaswami (1966)
6.	*Psathyrella subatrata* (Batsch) Gillet	3.2	Conical brittlestem	Kavanagh et al. (1952)

thermal degradation, microbial degradation, and chemical degradation (Abramowicz, 1990). Their degradation products are mostly chlorine-containing compounds that are potentially toxic.

Many reports on the consequences of POPs on industry and agriculture have been published in recent decades. POPs have improved pest and disease prevention, agricultural production, and industrial uses. During the post-World War II rise in rapid industrialization, many were widely used commercially, resulting in a wide geographic dispersion. Figure 6.9 highlights some of the POPs' sources (Chandra & Kumar, 2016). POPs are extremely stable in a wide range of conditions.

As a consequence of inadvertent formation, POPs compounds can be discovered in incineration, chemical industries, other fuel combustion, wildfires, putrefaction, and PCB-containing wastes. This type of waste can be found in several places and is the consequence of several activities such as the use of old oil, apparatus maintenance and repair, and building destruction (Ying et al., 2005; Wenzl et al., 2006). Pollutants, oil, fates, liquid fuels, dirt, ash, and silt enter the water system through wastewater from POP-producing or POP-using plants, as well as runoff from fields and roadways and atmospheric deposition. POPs gather up in the oceans and seas, which are their main reservoirs, due to river sediments, air pollution, waste disposal, and accidents. As seen in Fig. 6.9, POPs are retained in sediments on seacoasts, oceans, and huge lakes, where they are being released and re-enter the environment after a period of time (El-Shahawi et al., 2010; Galiulin et al., 2002).

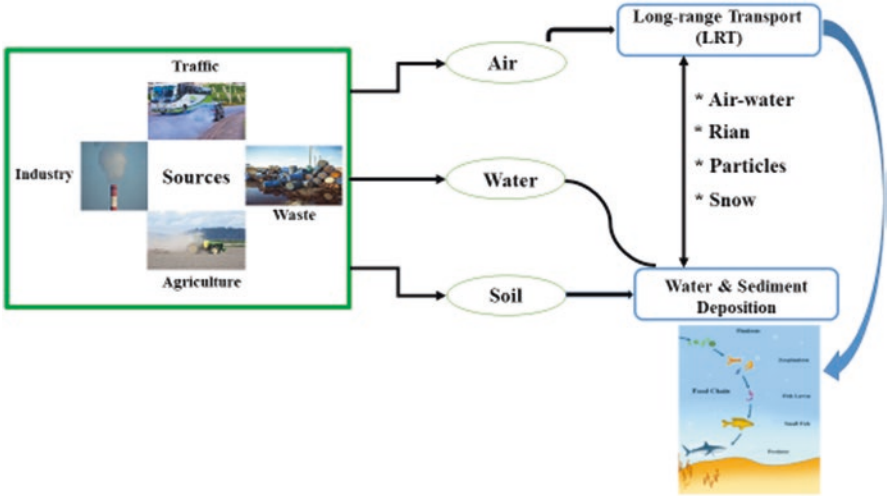


Fig. 6.9 Process during long-range atmospheric transport of POPs in the environment

6.1.6 *Distribution and Environmental Fate of Persistent Organic Pollutants (POPs)*

- POPs have an anthropogenic origin and are released into the atmosphere, waterways, and lands, where they accumulate in water and subsequently reach the food chain through sediment.
- Air and ocean currents distribute POPs over the earth, allowing them to travel a long distance.
- They then permeate weather events, air-water exchange, and phases like rain, snow, and suspended particles, exposing even some of the most remote human and animal populations that depend on marine nourishment.
- POPs can travel great distances and can be detected far from agricultural and industrial areas, such as the Northern Hemisphere.

The oceans cover 70% of the earth’s surface and contain significant organic carbon reserves. As a result, the oceans play a critical role in influencing the environmental transit, fates, and drains of POPs at regional and global levels (Wania & Mackay, 1996; Dachs et al., 2002), even though POP levels in the open ocean are lower than those found in coastal areas (Iwata et al., 1993; Dachs et al., 1997). Because of their huge oceanic volume, they may constitute a significant POP reserve (Jurado et al., 2004a, 2004b). Research on oceanic organic pollutants has traditionally focused on hydrophobic, persistent, and semi-volatile pollutants, such as POPs. Other organic contaminants, such as perfluorinated chemicals, have recently gotten a lot of attention (Jahnke et al., 2007). The pathway of atmospheric transport, deposition, and incorporation into plankton and transmission via the food web is critical for understanding the fate and transit of those chemicals that can undergo environmental transport due to their semi-volatility as all residual POPs (Pistochi & Loos, 2009). Rivers can be substantial contributors of POPs to the coastal environment, even for residual POPs (Gomez-Gutierrez et al., 2006; Zhulidov et al., 2000).

The most significant and fastest pathway for POPs to reach surface waters is through the atmosphere. POPs can be separated from the climate through four (4) different processes: (i) dry deposition of particle-bound pollutants, (ii) gaseous exchange between the atmospheric and the ocean's surface, (iii) wet deposition (rain scavenging), and (iv) degradation of hydroxyl (-OH) radical. Figure 6.8 depicts the main mechanisms that affect POPs in the oceanic atmosphere. Numerous studies have highlighted the importance of air-water exchange in understanding the environmental fates of POPs at local, regional, and global levels (Hornbuckle et al., 1995; Nelson et al., 1998; Totten et al., 2004; Rowe et al., 2007; Gioia et al., 2008a, 2008b; Nizzetto et al., 2008). A concentration difference stimulates the air interchange of pollutants between the climate and sea, while molecular and turbulent motion transports the contaminants. For those POPs (PCBs, HCHs, lighter PAHs) whose atmospheric occurrence is primarily in the gas phase, in comparison to wet and dry deposition, air-water gas exchange is the most dominant depositional mechanism in the world basis (Jurado et al., 2004a, 2004b).

Deposition, both dry and moist (Jurado et al., 2005), can have a substantial impact on some PAH and PCDD/F, which have a greater affinity for aerosol soot carbon due to their planar form (Dachs et al., 2000; Lohmann et al., 2007). Readily accessible measured values of POP air-water exchange fluxes in remote ocean water areas are severely limited due to the challenges related to POP sampling techniques, ship and laboratory pollutants, and the costs of using ships in the open ocean.

In general, the environment is a net supply of organic contaminants to aquatic bodies in the open ocean. In some parts of the Atlantic Ocean, Gioia et al. (2008a) found air-water fugacity ratios greater than unity. Lakaschus et al. (2002) proposed near-equilibrium conditions between the atmosphere and the ocean for less hydrophobic persistent organic pollutants, e.g., hexachlorocyclohexane (HCHs). Thus far, the only evident example of organic pollutant net volatilization flows has been recorded for lighter PAHs like phenanthrene, but this is presumably attributable to the in situ biogenic synthesis of this hydrocarbon. For legacy POPs like PCBs, Zhang and Lohmann (2010) described a PCB volatilization flux from the open water in the south Pacific. Volatilization of POPs is a common occurrence in coastal environments. The NE Atlantic Ocean and the NE Mediterranean Sea, for example, have shown high volatilization trends (Yan et al., 2008; Garcia-Flor et al., 2009). Polar POPs such as nonylphenols have shown similar volatilization fluxes in the North-East Atlantic and North Sea (Van Ry et al., 2000; Xie et al., 2006). Significant river systems and runoff fluxes into the marine environment could generate POP volatilization fluxes from coastal waters, and POPs may be transported from coastal waters to open sea regions.

The importance of the main routes for the transport of pollutants from coastal ocean to open sea waters has received little attention so far. When analyzing the air-water exchange, these pollutants have been implicitly examined for residual persistent organic pollutants which may be carried by grasshopping, i.e., successive volatilization and deposition. However, little research work was performed by the researchers on ocean currents, and mixing mechanisms in the water column move compounds with increased solubility and limited volatility in the marine environment.

It has been proposed that PFOS (perfluorooctanesulfonic acid) and PFA (perfluoroalkoxy) could act as inactive tracers (Yamashita et al., 2008), even though more research is needed to understand the process of degradation of a few compounds in the environment (Yamamoto et al., 2007).

They have been reported across every climatic segment, including in places where POPs were never created or used, such as the open sea and the polar regions. As a consequence, atmospheric transport has been recommended as the major route for spreading semi-volatile chemicals thousands of kilometers away from crowded areas. POPs, also known as persistent bioaccumulative toxic chemicals (PBTs), are bioaccumulated substances with long-term persistence in the environment and the ability to travel large distances through the atmosphere (Fig. 6.10).

Within the last few decades, anthropogenic POP imports into the marine ecosystem have increased significantly. High quantities of varying dangerous POPs, including pesticides and heavy metals, are frequently reported in estuaries, coastal areas, and “enclosed” seas like the Mediterranean Sea (Di Bella et al., 2006; Ansari et al., 2004). POPs are the most well-known source of contamination; they are primarily synthetic compounds, which can bioaccumulate in organisms after accumulating in the environment for a long time (Gui et al., 2014), contaminating foodstuffs, including those that have not been treated directly (Panseri et al., 2014).

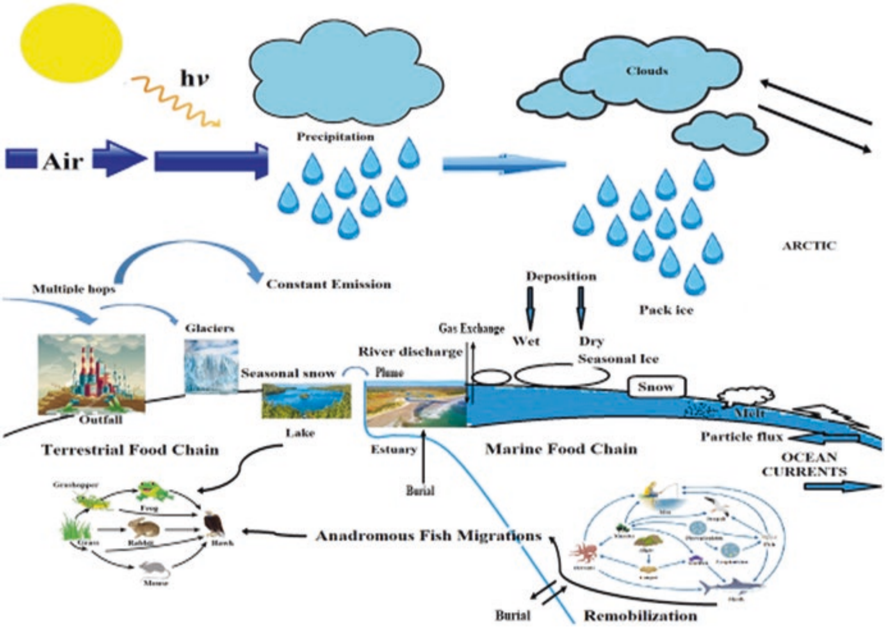


Fig. 6.10 Environmental fate of persistent organic pollutants (POPs)

Understanding the sources, routes, and fate of purposely created POPs requires documentation of their initial synthesis and formulations, diffusion, usage, and disposal. The sustainability of many contaminants is seen in the importance of soils and sediments as ongoing sources (productive agricultural implementations, polluted garbage, industry, and harbor sites) (Hermanson & Hites, 1989; Jantunen et al., 2000). On global and regional dimensions, exchange across distinct environmental categories is an essential component of describing the POP cycle once released into the environment (Hornbuckle et al., 1994; Swackhamer et al., 1998; Macdonald et al., 2000). Furthermore, understanding the anthropogenic and natural mechanisms which produce a few other persistent organic pollutants (viz., dioxins and furans) provides a framework for determining how these chemicals infiltrate the environmental resources. This knowledge provides the baseline for risk assessment when combined with the chemical characteristics of the POPs in concern and the nature of environmental processes influencing their mobility. Given the "ubiquitous" distribution of many POPs, this study of dispersal pathways is important for both men and wildlife.

1. *Global Distribution:* POP dispersion in the global ecosystem is known to be influenced by atmospheric dispersal. The influence of POPs' physicochemical features on their atmospheric spread to distant, cold polar locations of the earth has been studied using global models (Wania & Mackay, 1995; Wania & Mackay, 2001). As transport and fate models, formalized methodologies like the Quantitative Water Air Sediment Interaction (QWASI) fugacity model created for Lake Ontario and mass-balance analyses of specific basins and watersheds can be used (Mackay, 1989; Eisenreich et al., 1981; Hornbuckle et al., 1994; Hoff et al., 1996). Data from across the system has produced thorough overviews of Arctic contamination. These include abiotic and food chain evaluations, transect-based investigations of pollutants in sea waters and samples of air, and comprehensive international assessments (Muir et al., 1992, 1999; Iwata et al., 1993; CACAR, 1997; AMAP, 1998).
2. *Environmental Compartmentalization*

   The physicochemical characteristics of certain substances, like as the octanol/water coefficient (Kow), can be utilized to forecast biomagnification across the food web. PCBs that are "dioxin-like" and have a log Kow of >4.0 are highly bioaccumulative and persistent in food systems (Shaw & Connell, 1984; Oliver & Niimi, 1988; Hawker & Connell, 1988).
3. *Metabolic Degradation:* It is necessary to analyze organisms' metabolic capacity to target some compounds, resulting in either detoxification and removal or the creation of toxic metabolites. For polychlorinated biphenyls (PCBs), congeners with adjacent unsubstituted pairs of C-positions on the phenyl (C_6H_5-) rings are more susceptible to metabolic attacks and elimination via the generation of soluble metabolites (Boon et al., 1997). The capability of species (such as aquatic animals) to remove various types of polychlorinated biphenyls should help with risk evaluation by differentiating between persistent and less persistent compounds, as well as the stimulation of detoxifying enzymes in the hepatic tissues

(Boon et al., 1994; Kannan et al., 1995). In mammals, however, the synthesis of hydroxy-polychlorinated biphenyls and other persistent organic pollutant metabolites ("reactive intermediates") as a result of enzymatic activities may play a crucial part in the breakdown of some endocrine systems (Brouwer et al., 1998; Sandau et al., 2000). Documenting pattern changes as a result of these metabolic processes at each level of the food chain is critical to assessing human and wildlife exposure in terms of transportation and fate.

4. *Degradation Processes:* There are two factors (abiotic and biological) that can cause POPs and "dioxin-like" chemicals to degrade slowly (Abramowicz, 1995). There is a lot of diversity between compounds degradation under different conditions. Sedimentation is a process that reduces the biological availability of pollutants in the food web and behaves as a "sink." Despite these processes, numerous persistent organic pollutants and "dioxin-like" compounds persist in the climate, which is a hallmark of POPs and justification for an integrated risk assessment.

When POPs are emitted, they split into gas and aerosol phases which makes them vulnerable to long-range atmospheric transport (LRAT). Atmospheric depositional mechanisms are very important in the transport and fate of persistent organic pollutants on a regional and global level. Then, we can apply any one of the three primary processes for the deposition of pollutants:

1. Dry particulate-bound pollutant depositing.
2. Between the atmospheric boundary layer and diffusive gas exchange (ABL).
3. Transport of semi-volatile organic compounds from the atmosphere to the ocean occurs *via* the surface ocean and rain scavenging (either from gas or particle phases) (Jurado et al., 2004a, 2004b).

Climate change has already prompted a plethora of environmental challenges, and contaminant emissions, dispersion, and toxicity are all linked to changes in climate. Both the IPCC assessment and the UN Environment Program's (UNEP) annual report emphasized the need of paying attention to environmental degradation, particularly in the light of global warming (Climate Change, 2013). The POP series has already been covered, and Fig. 6.10 depicts the transport pathways for persistent organic pollutants. POPs travel around the world in two ways: through atmospheric circulation and ocean currents. POPs can persist in the atmosphere in both gaseous and particle forms. As a consequence of atmospheric circulation, persistent organic pollutants in the environment can be dispersed worldwide via gaseous and particle transport. When the temperature decreases, POPs are more likely to be precipitated on the ground surface; when the temperature rises, however, they evaporate and migrate back into the environment.

This cycle is repeated, allowing POPs to be transferred and stored in remote locations. It is referred to as the grasshopper effect. Moreover, some persistent organic pollutants that have a more solubility, namely, hexachlorocyclohexanes (HCH) and perfluorooctane sulfonates (PFOS), can penetrate waterbodies, flow into ocean circulation, and migrate all around the world. Changes in climatic parameters such as temperature, wind speed and direction, and precipitation are all caused by climate change. As these conditions change, the intensity and paths of POPs delivered by air and ocean will certainly transition (Wang et al., 2016).

Long-Range Environmental Transport

POPs are distributed globally as a result of a combination of local, regional, and long-distance travel. POPs can be detected in remote areas due to their ability to undergo long-range environmental transport (LRET), which is facilitated by their persistence. The dispersion of organochlorine pollutants was first documented in the 1960s, and public concern about OCPs sparked the "Silent Spring" revolution; it sparked a global social movement based on research to raise chemical environmental awareness.

Persistent organic pollutants may be distributed in nature through the air, water, or a mixture of air and water (land and water matrices are referred to as reservoirs). The existence of persistent organic pollutants in both air and water compartments is a cause for concern, as bioconcentration and subsequent biomagnification through trophic chains are the two main pathways of bioconcentration, posing a risk to humans and wildlife. Furthermore, in these highly dynamic media, pollutant dispersion is expedited and amplified.

Physicochemical qualities and environmental variables influence a contaminant's possible fate and behavior. The "grasshopper effect" has been described as a phenomenon in which mobilization happens in pulses of precipitation and volatilization. POPs may be subjected to various degradation and/or accumulation processes (e.g., photolysis, biodegradation, soot sequestration, or bioaccumulation) at various stages of the cycle, affecting their presence, fate, toxic effects, and bioaccumulation. The grasshopper effect is primarily controlled by heat and the biological process. It is a worldwide process of distillation aimed at the icy poles, where persistent organic pollutants have been found to accumulate. Globally, the pattern of POPs' distribution depends on many other environmental conditions which include a source of origin, deposition, absorption of organic matter, and many more secondary sources.

These POPs become persistent across the planet because they are photolytically, biologically, and chemically resistant and volatile. Every continent, including the open oceans, deserts, the Arctic (Iwata et al., 1994; Barrie et al., 1992) and the Antarctic reported maximum level of POPs (Tanabe et al., 1982; Bidleman et al., 1989).

A substance's volatility and air temperature, in general, influence its dispersion pattern. As a consequence, highly volatile POPs tend to travel from hot and temperate to colder environments. The "grasshopper effect" characterizes the distribution pattern of persistent organic pollutants to higher latitudes, which consists of a series of small bounces in which they pause and rest at mid-latitudes if the seasonal temperature varies and then continue to travel when the temperature is appropriate (Wania & Mackay, 1996). Each POP has its own unique ambient condensation temperature or temperature range. Due to their volatile nature, these POPs can easily reach the polar regions by air. As the result, they have been dispersed throughout the world depending on the degree of volatility, and this type of distribution is known as global fractionation. Even if there is a decrease or pause in their release, these toxins will continue to pollute the environment as they are expelled from deposited chemical reservoirs.

Wania and Mackay (1996) reported that the occurrence and retention of POPs in polar regions are due to the global distillation/fractionation effect. Pesticides applied in hot climates may evaporate and move to cooler climes, where they might condense at high concentrations. This "cold wall rule" demonstrates the widespread presence of organochlorines in the Arctic and Antarctic fauna. Goldberg et al. (1975) used the terminology "global distillation" to describe the migration of dichlorodiphenyltrichloroethane from continents to marines. This process is also known as "cold condensation," "cold finger," or "cold trap" (Ottar, 1981). According to the latest study, the chronic spatial distribution patterns of OCPs have increased recently as a result of the substances' physicochemical properties and some of the unique characteristics of colder climates.

The global distribution of POPs is caused by a combination of local, regional, and long-distance transportation. Because of their ability to undergo LRET, POPs can be detected in remote regions, which is assisted by their persistence

As a result, the overall picture of the system and their interrelationships in the global ocean is extremely complicated, and it is influenced by a variety of environmental conditions and multi-media equilibria.

Conclusion and Future Prospective

This chapter aims to provide an updated comprehensive overview of the occurrence, distribution, and fate aspects of persistent organic pollutants in the environment. In addition, the chemical structures and characteristic features of specific POPs are discussed. Organohalogenated chemicals, polycyclic aromatic hydrocarbons, and pesticides are examples of POPs that can be found in a variety of environmental compartments. Their existence in the aquatic environment is a global issue, with sediments serving as a storehouse and, as a result, a source of hydrophobic, persistent, and toxic substances.

Pollution by POPs is pervasive, and it travels around the world through the air, sea, and other routes. In terms of persistence, long-range transport (LRT), bioaccumulation, and toxicity, the Stockholm Convention adopted requirements for new POP candidates. As countries became aware of the risks of POPs, they began to limit their manufacture, usage, and release in the environment. This legally binding global agreement aims to decrease and eliminate the discharge of 12 POPs, including pesticides and industrial chemicals, as well as POPs created accidentally. POPs have a variety of harmful impacts and can affect the function of nearly every organ. The working of these compounds is mediated by a variety of mechanisms, including enzyme induction, endocrine action, alteration in gene regulation, and changes in the nervous system's function.

As a result, the presence of persistent organic pollutants in our climate is a worldwide concern and a challenge on a global level. In general, research on POPs is limited, and more research is needed to expand the knowledge in this field. It is now clearly obvious how harmful POPs can be in the future for coming generations. It has an impact not just on the environment, but also on wildlife animals and, directly or indirectly, on humans. Persistent organic pollutants are becoming a big concern as a result of increased industrial use, producing their accumulation and persistence in living organisms and the surroundings. Due to many pathways, exposure to POPs has numerous lethal consequences for organs and tissues, including oxidative stress and cell death. POPs have been the subject of several international programs to restrict their use and eventually eliminate them.

In this regard, further research and development need to be carried out on POPs regarding the side effects of their application, which should always be prioritized for the welfare of mankind.

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Chapter 7

Phyco-remediation: Role of Microalgae in Remediation of Emerging Contaminants

Saket Jha, Rohit Shukla, Ravikant Singh, Mohee Shukla, Piush Srivastava, Anand Pandey, and Anupam Dikshit

Abstract The developing new technologies and urbanization make ease to livelihood but also create new challenges to mankind and living creatures. There are several toxic compounds rigorously released in the environment, and due to their high hydrophobic nature as well as their non-degradable properties, they become a major concern to human health. There are several techniques introduced to treat these emerging contaminants like endocrine-disrupting compounds, pharmaceutical compounds, antiepileptic drugs, antibiotics, dyes, wastewater contaminants, organic and household wastes, and personal care products. The remediation processes are introduced using plants, fungi, and bacteria. The algae such as microalgae and cyanobacteria are found more reliable and good biological tool to remediate these contaminants. The process of removal of these emerging contaminants using algae is called phyco-remediation. New technologies like genetic engineering incorporated with these microalgae are a promising tool to cope up with these emerging contaminants and help in detoxifying these contaminants from the ecosystem. The present chapter discusses all aspects related to the concept of emerging contaminants and its types and the role of microalgae and cyanobacteria in the removal of these contaminants with proper mechanisms. Also, this chapter provides some future prospects

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S. Jha (✉) · M. Shukla · A. Dikshit
Biological Product Laboratory, Department of Botany, University of Allahabad, Prayagraj, India

R. Shukla
Centre of Science and Society (under IIDS), University of Allahabad, Prayagraj, India

R. Singh
Department of Biotechnology, Swami Vivekanand University, Sagar, Madhya Pradesh, India

P. Srivastava
College of Medicine, University of Illinois, Chicago, IL, USA

A. Pandey
Department of Botany, Avvaiyar Government College for Women, Karaikal, Puducherry, India

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regarding the applications of phyco-remediation with respect to some other fields like advanced oxidation processes and genetic engineering.

Keywords Phyco-remediation · Emerging contaminants · Endocrine-disrupting compounds · Pharmaceutical compounds · Antiepileptic drugs

7.1 Introduction

Environmental toxicities like heavy metals, nanomaterials, micro-wastes, and emerging contaminants are now a major concern as these are not easily degraded. These can persist for thousands of years and cause severe diseases as well as chemically toxify the soil, water, and air. Chemicals such as pesticides, pharmaceutical wastes, and household products are considered emerging contaminants (Maryjoseph & Ketheesan, 2020). These emerging contaminants (EC) are now causing hazardous effects on the aquatic environment (Bilal et al., 2019). The common emerging contaminants are pharmaceuticals by-products, personal care products (PPCPs), endocrine disruptors, surfactants, plasticizers, etc. (Nguyen et al., 2021). In these ECs, the most frequent and dominant ones are pharmaceuticals, personal care, and household products as these are increased up to approximately 9–10 thousand tons per year (Wilkinson et al., 2017; Maryjoseph & Ketheesan, 2020). These have now become a prominent concern for the environment as these are non-biodegradable as well as biologically accumulated which create the toxicity to aquatic bodies and soil. This indirectly or directly affects human health (Ummalyma 2020). Pesticides and industrial organic waste products directly affect the vertebrate's endocrine system which causes the deformation in fishes and all mammalian classes (Matamoros et al., 2015). Several compounds from personal care products cause skin problems mainly in humans (Xiong et al., 2018; Maryjoseph & Ketheesan, 2020).

Several international environmentalists, ecologists, and concerned departments are taking these ECs with high priorities, and some departments have given major directive guidelines for the regulations of these emerging contaminants (Norvill et al., 2016; Maryjoseph & Ketheesan, 2020). Till now, these ECs are still not remediated as well as treated properly in sewage treatment plants or other resources/sites (Maryjoseph & Ketheesan, 2020), and several new methodologies are designed to treat these ECs (Ali et al., 2018). But still, these techniques are not sufficient to completely degrade or remove these emerging contaminants in wastewater resources (Sutherland & Ralph, 2019).

Several species of algae are now being used to remove these contaminants (Olguín, 2003). From these phycobionts, microalgae are widely used for remediation of these contaminants as they have good potential to degrade these toxic components (Agüera et al., 2020). There are several types of research and literature that provide the evidence and potential effectiveness of these microalgae for degrading the emerging contaminants from soil and wastewater treatments (Sutherland & Ralph, 2019). Nowadays, researchers are focusing on these microalgae in the removal of other contaminants like pharmaceutical residue and industrial wastes (Sakarika et al., 2020). Various techniques are also introduced using these phycobionts in the degradation of these emerging contaminants. These remedial investigations using microalgae are found to be cost-effective and simpler methods in wastewater management (Maryjoseph & Ketheesan, 2020).

7.2 Types of Emerging Contaminants

There are several types of emerging contaminants which are mentioned below (Table 7.1 and Fig. 7.1).

7.2.1 Wastewater Contaminants

The discharged contaminants from houses, farms, and industries are considered as wastewater contaminants, and these are mostly organic and inorganic chemicals which further react together to form more toxic compounds and thus adversely affect the environment and ecosystem (Ahmed et al., 2022). Without proper treatment, these contaminants have potential to affect human health along with environmental destruction, and it is an utmost priority to have a program around the globe to ensure the availability of quality water (Abdel-Raouf et al., 2012). Many organic and inorganic substances are present in wastewater (Goswami et al., 2021). Waterborne pathogens can cause several diseases to humans, and this is alarming for mankind (Wen et al., 2020). A hike in biochemical oxygen demand (BOD) might be noticed due to mixing of untreated wastewater in freshwater bodies (Siddiki et al., 2021). To reduce the pollutants and for the treatment of wastewater, several plants are established in cities and rural areas. These plants are based on the three basic levels of processes, namely, physical or mechanical which removes the stones, pebbles, etc., biological or microbial, and the last is chemical for the reduction of organic waste and pathogens (Eerkes-Medrano & Thompson, 2018). These urban treatment plants fail to remove the pesticides, pharmaceuticals, and other harmful materials (Grady Jr et al., 2011; Ahmed et al., 2020).

7.2.1.1 Degradation Challenges of Emerging Contaminants in Wastewater Treatment

The limitation with wastewater plants creates the major problem in removing these emerging contaminants which come mainly from industries, houses, factories, etc. (Table 7.1, Figs. 7.1 and 7.2). There is a need to improve more on treatments of these effluents (Oulton et al., 2010). There is also no proper plan to execute these contaminants as conventional treatments have limitations. Basically, conventional treatments are used to treat the BOD degradation but fail to degrade the high

Table 7.1 List of the emerging contaminants and remediating algae

Emerging contaminant type	Compounds	Name of algae	References
Pharmaceuticals	Ibuprofen	*Chlorella* sp., *Scenedesmus* sp., *Phaeodactylum tricornutum*, *Coelastrum* sp.	Villar-Navarro et al. (2018), Mulla et al. (2019)
	Diclofenac	*Chlorella sorokiniana*, *C. vulgaris*, *Scenedesmus obliquus*, *Coelastrum* sp.	Escapa et al. (2016), Villar-Navarro et al. (2018), Mulla et al. (2019)
	Naproxen	*Cymbella* sp., *Scenedesmus quadricauda*	Ding et al. (2017), Mulla et al. (2019)
	Levofloxacin	*Chlamydomonas mexicana*, *C. pitschmannii*, *Chlorella vulgaris*, *Ourococcus multisporus*, *Micractinium resseri*, *Tribonema aequale*, *Scenedesmus obliquus*	Xiong et al. (2017a, 2017b), Mulla et al. (2019)
	Carbamazepine	*Pseudokirchneriella subcapitata*	Vernouillet et al. (2010), Mulla et al. (2019)
	17α-Ethinylestradiol and 17β-estradiol	*Chlorella* sp., *Nitzschia acicularis*, *Selenastrum capricornutum*, *Chlamydomonas reinhardtii*	Sole and Matamoros (2016), Parladé et al. (2018), Hom-Diaz et al. (2015), Mulla et al. (2019)
	17α-estradiol and estrone	*Scenedesmus dimorphus*, *Chlorella vulgaris*	Zhang et al. (2014), Lai et al. (2002), Mulla et al. (2019)
	Enrofloxacin	*Scenedesmus obliquus*, *Chlamydomonas mexicana*, *Chlorella vulgaris*, *Ourococcus multisporus*, *Micractinium resseri*	Xiong et al. (2017c), Mulla et al. (2019)

Antibiotic	Tetracycline, florfenicol, metronidazole	*C. vulgaris*	de Godos et al. (2012), Song et al. (2019), Hena et al. (2020), Xiong et al. (2021)
	Enrofloxacin	*Chlorella vulgaris*, *Scenedesmus obliquus*, *Chlamydomonas mexicana*, *Ourococcus multisporus*, *Micractinium resseri*	Chen et al. (2020), Xiong et al. (2017d), Xiong et al. (2021)
	Amoxicillin	*Microcystis aeruginosa*	Liu et al. (2015), Xiong et al. (2021)
	Ofloxacin	*Chlorella sorokiniana*, *Scenedesmu obliquus*, *Desmodesmus* sp., *Coelastrella* sp., *Coelastrum astroideum*, *Chlorella vulgaris*	Gojkovic et al. (2019), Xiong et al. (2021)
	Azithromycin, lomefloxacin, roxithromycin, sulfamethoxazole, sulfamerazine, sulfamonomethoxine, clarithromycin, trimethoprim	*Haematococcus pluvialis*, *Selenastrum capricornutum*, *Scenedesmus quadricauda*, *Chlorella vulgaris*	Kiki et al. (2020), Xiong et al. (2021)
	ciprofloxacin, sulfadiazine	*Chlamydomonas*	Xie et al. (2020), Xiong et al. (2021)
	Ceftazidime, cefradine	*Chlorella pyrenoidosa*	Yu et al. (2017), Chen et al., 2015), Xiong et al. (2021)
Hydrocarbon	14c Naphthalene	*Agmenellum quadruplicatum*	Cerniglia et al. (1979), Gupta et al. (2015)
	R-endosulfan and its oxidation product endosulfan sulfate	*Chlorococcum* sp., *Scenedesmus* sp.	Sethunathan et al. (2004), Gupta et al. (2015)
	Fluoranthene, pyrene	*Chlorella vulgaris*, *Scenedesmus platydiscus*, *Scenedesmus quadricauda*, *Selenastrum capricornutum*	Lei et al. (2007), Gupta et al. (2015)
	Polyurethane	*Prototheca zopfii*	Ueno et al. (2008), Gupta et al. (2015)

(continued)

Table 7.1 (continued)

Emerging contaminant type	Compounds	Name of algae	References
Wastewater	Municipal wastewater, textile wastewater, dairy wastewater,	*Chlorella vulgaris*, *Botryococcus terribilis*	Cabanelas et al. (2013), Cuellar-Bermudez et al. (2017)
	Settled and activated sewage, animal wastewater, synthetic wastewater	*Chlorella pyrenoidosa* and *Scenedesmus* sp.	Tam and Wong (1989), Cuellar-Bermudez et al. (2017)
	Poultry litter wastewater, swine wastewater, olive oil mill wastewater	*Spirulina platensis*	Markou et al. (2018), Mezzomo et al. (2010), Cuellar-Bermudez et al. (2017)
	Brewery effluent	*Scenedesmus obliquus*	Mata et al. (2012)
	Piggery wastewater	*Ourococcus multisporus*, *Nitzschia* cf. *pusilla*, *Chlamydomonas mexicana*, *Scenedesmus obliquus*, *Chlorella vulgaris*, *Micractinium reisseri*	Abou-Shanab et al. (2013), Cuellar-Bermudez et al. (2017)
	Dairy wastewater, domestic wastewater, slaughterhouse wastewater, carpet mill	Consortia microalgae	Chinnasamy et al. (2010), Cuellar-Bermudez et al. (2017)

Pesticide	DDT (1,1,1-trichloro-2,2-bis (p-chlorophenyl) ethane)	*Anabaena*, *Nostoc*, and *Chlorococcum* spp.	Megharaj et al. (2000), Gupta et al. (2015)
	Fenamiphos (ethyl 4-methylthio-m-tolyl isopropyl phosphoramidate)	Five different species of cyanobacteria and green algae	Cáceres et al. (2008), Gupta et al. (2015)
	DDT, dieldrin, and photodieldrin	*Ankistrodesmus amelloides*	Neudorf and Khan (1975), Gupta et al. (2015)
	Organo-phosphorus pesticide, fenamiphos (ethyl 4-methylthio-m-tolyl isopropyl phosphoramidate) and its metabolites	*Chlorococcum* sp., *Pseudokirchneriella subcapitata*	Cáceres et al. (2008), Gupta et al. (2015)
	Monocrotophos, quinalphos	*Chlorella vulgaris*, *Scenedesmus bijugatus*	Megharaj et al. (1987), Gupta et al. (2015)
Dyes	Rhodamine B	*Coelastrella* sp.	Baldev et al. (2013), Singh et al. (2020)
	Azo dyes	*Chlorella and Oscillotoria*, *Spirogyra* sp., *Aphanocapsa elachista*	Mohan et al. (2004), Soundharya et al. (2016), El-Sheekh et al. (2018), Singh et al. (2020)
	Textile wastewater, indigo textile dye, tectilon yellow 2G	*Chlorella vulgaris*	El-Kassas and Mohamed (2014), Cheriaa et al. (2009), Acuner and Dilek (2004), Singh et al. (2020)
	Triphenylmethane dye, malachite green	*Cosmarium* sp.	Dhaneshvar et al. (2007), Singh et al. (2020)
	Blue and green colored dyes	*Chlorella vulgaris*, *Sphaerocystis schroeteri*	Raymond and Kadiri (2017), Singh et al. (2020)
	Methyl red and orange II,	*Nostoc linckia*, *Oscillatoria rubescens*	El-Sheekh et al. (2009), Singh et al. (2020)

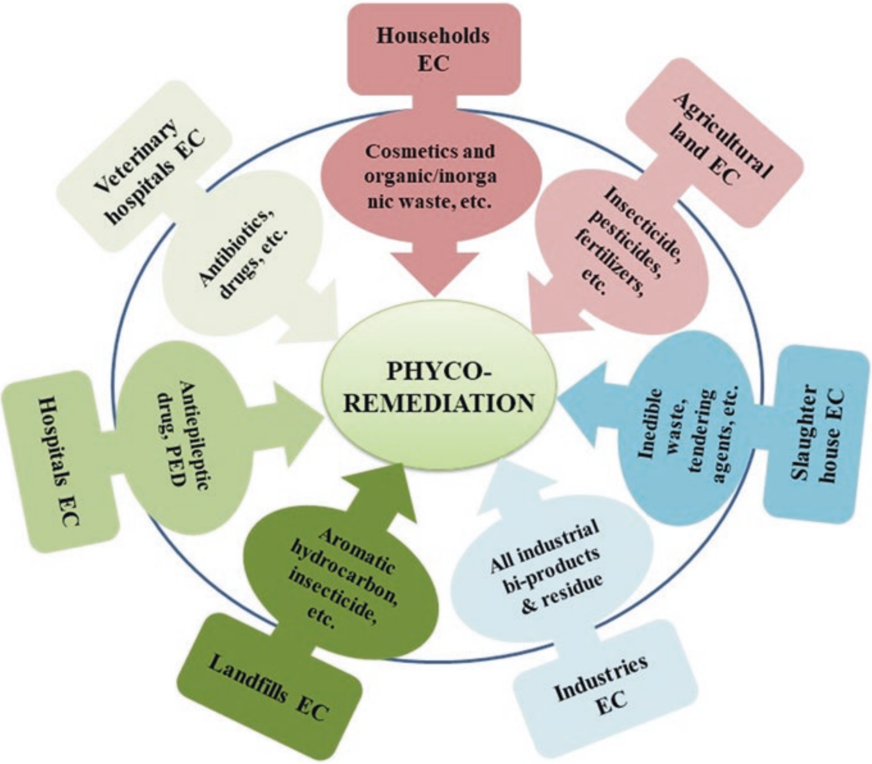

Fig. 7.1 Types of emerging contaminants and their sources

chemical stabilized compounds as well as low biodegradable contaminants. Sometimes these contaminants react with diverse compounds which makes them more stable and variable during the treatment process (Gupta et al., 2015).

The major problem in the treatment of these contaminants is not having appropriate and not well-designed procedure to detect as well as quantify these ECs at their lowest level (Battaglin & Fairchild, 2002; Petrović et al., 2003). Due to this reason, these ECs are still a major concern to degrade, and there is always a need to develop the proper conventional protocol (Gupta et al., 2015). Generally, these ECs have low biological degradation activity with conformist microbial populations present in the sewage in its natural environment. This results in ineffective removal of these ECs. The biggest problem in degradation of these contaminants using biological entity is due to the lack of rich microbial inoculants in treatment plants (Kasprzyk-Hordern et al., 2009). Most of the pharmaceutical constituents are highly water-soluble and readily react with water to form more complex and toxic compounds (Petrie et al., 2015).

Another significant characteristic of these compounds is its segregating between compact and fluid phase liable on their adsorption factors and repellent property

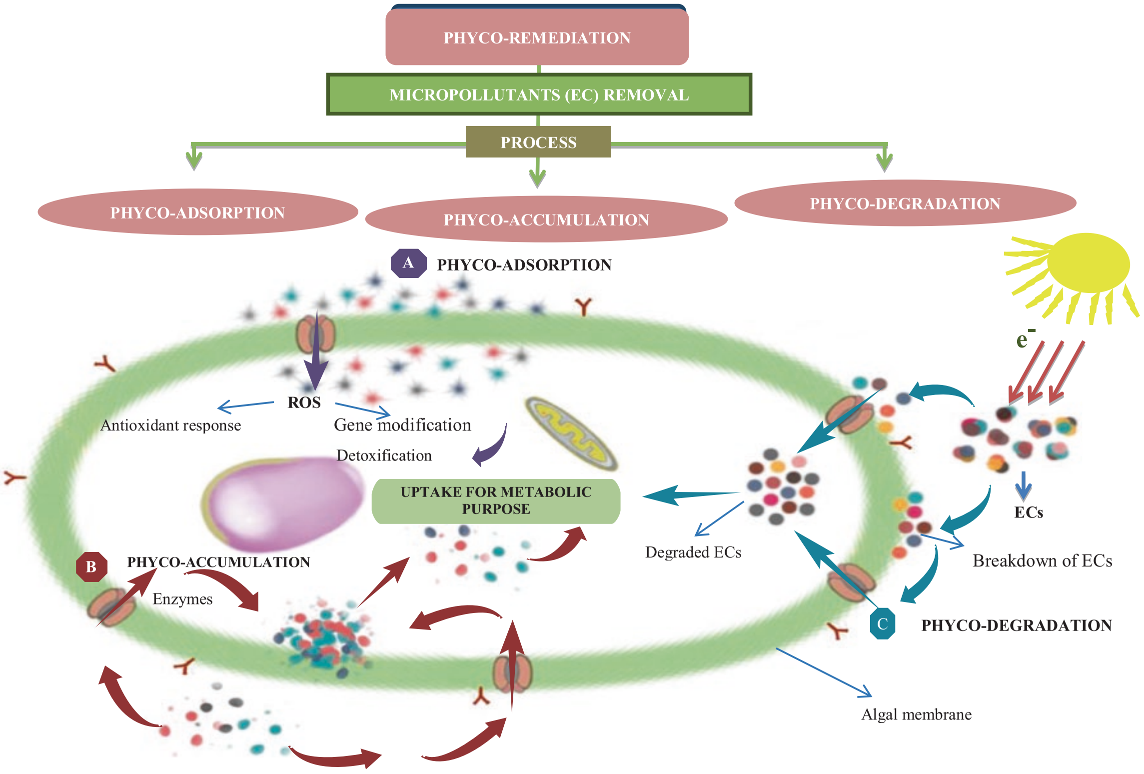

Fig. 7.2 Mechanisms showing phyco-remediation of emerging contaminants (ECs)

with water molecules. Several organic-cidal compounds (pesticides, insecticides, herbicides, etc.) are highly water repellent and thus resist in their solid phases instead of degrading underwater (Petrie et al., 2015). pH plays an important role in complexity of these contaminants, and various other factors regarding soil widely vary with these ECs (Petrie et al., 2015). These contaminants are not easily broken in soil and remain intact in their solid structures due to their high resistivity against water molecules (Hyland et al., 2012; Gupta et al., 2015).

7.2.1.2 Ecological Fate of Emerging Contaminants

These compounds are generally discharged in the environment because they are not properly treated as well as processed. For example, waste of houses and hospitals are well regulated before being discharged to the environment. These are moved directly or indirectly in water bodies via rain or various means. These contaminants continuously react with each other as well as surrounding molecules/compounds to become more toxic to the accumulating environment. After becoming more stable, these are not biologically degradable and abruptly affect the surrounding environment (Gupta et al., 2015).

These contaminants reach the water reservoirs such as river, pond, and seas. Majority of these are untreated and released directly. These caused serious threat to aquatic organisms and sometimes led to biomagnification process and thus caused various diseases to humans. These contaminants also showed photolysis in aquatic environment by the presence of microbial activity mainly by algae and cyanobacteria (Lin & Reinhard, 2005). Though, capability of the contaminants further degraded via photolysis, and may variable due to their chemical composition and structure. These degradations also depend on microbial capabilities. But, their degradation pathway is still unknown as continuously several researchers are still searching these pathways (Gupta et al., 2015).

The various photolysis done by indirect pathway is processed by microbes releasing these free radicals (Ryan et al., 2011). These toxic components do not degrade easily, persist and undergo biological accumulation in organisms, and thus lead to biological degradation by these microbial inoculants. The engulfing of these emerging contaminants abruptly affects the biological system and also indirectly or directly adversely affects the ecosystem (Gupta et al., 2015). Several emerging contaminants which are hydrophobic get absorbed with other organic solids and only degrade in liquid phase procedure.

However, these contaminants' degradation pathways are still limited and need more emphasis on these processes. Due to not available procedures for their detection and quantificational analysis, they cause severe threat due to their future accumulations. Since, many intermediate compounds are found to be more toxic than their parental components, this makes them more complex and vulnerable to degradation processes of these complex contaminants (Gupta et al., 2015).

7.2.1.3 Eco-toxicological Risks of Emerging Contaminants

The alarming concerns towards these emerging contaminants are due to their serious threat to all living creatures including humans. The majority of these ECs reflects the acute as well as chronic diseases which completely depend on their concentrations in the ecosystem and level of exposure (La Farre et al., 2008). These ECs mainly mimic the similar hormonal effect as the organismal hormones behaviors and thus lead to causes endocrine-disrupting activities. These contaminants cause severe endocrine interference even in lower amount as compared to concentration at which normal enzymes perform their activity (La Farre et al., 2008). Complexity in their structure and behavior is another factor which makes them more toxic components (Cleuvers, 2004). The complexity and toxicity of these contaminants was investigated by Petrie et al., (2015), in which they found that the different chiral enantiomers of these compounds play a vital role in their variation of toxic nature and resistant to biodegradation. These reports are earlier produced by several researchers also, and due to this, researchers introduced the noble applications to remove these contaminants with their overall toxicity level (Gupta et al., 2015).

7.2.2 Pesticides

Humans are known to use pesticides since ancient times and mainly used arsenic and fluoride compounds to cope up with several pests (Battaglin & Fairchild, 2002). After positive responses from these pesticides, several other organic compounds are introduced to be used as pesticides such as DDT (dichlorodiphenyltrichloroethane). These pesticides are used widely and uncontrolled by farmers and thus remain persistent in soil and aquatic environment. Now, these are accumulated in the ecosystem in such a huge amount that these become a serious threat to cause various diseases to human health. Several studies are following the removal of these pesticides (Table 7.1, Gupta et al., 2015). After imposing a ban on the utility of these pesticides, still these pesticides are used till today and show an adverse effect due to their presence in the ecosystem. At very low concentrations, they show carcinogenic activity on human tissue as well as act as an endocrine-altering agent. Several researchers reported about their toxicity effect on human cell lines, and several other new pesticides are also reported to have similar hazardous effect (Gupta et al., 2015). However, rigorous investigations provide evidences about pathways and their disease-causing properties. This also helps in providing their mechanism of causing toxicity in organism and provides for understanding the ability of removing these toxic pesticides. There is still more to be known about their metabolic activities and their interactions with other compounds, and this makes these pesticides as one of the major concerns for their biological remediation process (Gupta et al., 2015).

7.2.3 Endocrine Disruptors

There are several natural and synthetic compounds which exhibit similar properties (biological and chemical) like hormones, and due to this reason, these compounds interfere with the normal function of organisms (Gupta et al., 2015). A vast range of compounds have been demonstrated as having properties to interfere with the endocrine system henceforth named as endocrine-disrupting compounds (EDC). Pesticides, dioxins, synthetic hormones, and surfactants are some examples of EDC (Richardson & Ternes, 2018; Gupta et al., 2015). EDC possess a great affinity with different types of hormonal receptor and pathways which makes them more complex, stable, and variable contaminants. Organisms face harsh effect on their ecological system due to continuous exposure to these compounds (Gupta et al., 2015).

7.2.4 Pharmaceutical Products

A majority of the emerging contaminants are introduced by pharmaceutical residues such as antibiotics, medicinal drugs, compound-based adhesives, and antidepressants, which are directly discharged to water resources (Kuch & Ballschmiter, 2000). Due to their increasing amount in water bodies and causes serious threat to human health as well as aquatic organisms makes it great concern for the removal of these contaminants. This leads to the biggest challenge for researchers and authorities to degrade these contaminants using a cost-effective and environmental-friendly method (Gupta et al., 2015). Extreme investigation and novel applications are now seriously demanded to remove or break down these pharmaceutical residues from the ecosystem especially in water bodies. These compounds exert lethal effect on all living creatures of earth directly or indirectly as these are combined with other compounds. Researchers are still focusing on the exact mode of action and abrupt effect of these contaminants on cell lines in the ecosystem, and their mechanism is still needed to be understood. This is the only concern to provide a clear understanding for removal or degradation of these pharmaceutical residues (Webb, 2001).

The level of toxicity of single drug is found to be lesser than the combination of many drugs (Cleuvers, 2003). These pharmaceuticals drugs when mixed together showed higher toxicity. This study provides the idea to detoxify these combined drugs. Microalgae here find good source to remediate these contaminants. But, still it becomes major concern as these are in large quantity and not easily detoxified by algal as well as other means of remediation processes (Cleuvers, 2003). Wilson et al. (2003) reported that these algae are growing very fast on these contaminants and adopt their new food habits. Also, their cellular structure changed from non-treated algal biomass (Gupta et al., 2015).

7.2.5 *Personal Care Products*

The ecological hazardous status of personal care products (PCPs) are never taken in serious concern. Various compounds like triclosan (TCS) are being used in making personal care products and used widely by consumers since the last three to five decades (Katz et al., 2013). These consumed products are released in water reservoirs without being treated properly and make the aquatic ecosystem more toxic and contaminated (Gupta et al., 2015). Wastewater treatment plants and sewage treatment plants are major sources of these contaminants which get discharged directly into rivers and seas without regulatory processes. This is because these plants do not have proper regulatory systems or methods to remove these toxicities from wastewater (Kumar & Xagoraraki, 2010; Fernandes et al., 2011).

Earlier reports found that these PCPs are hazardous and cause severe effect on health of aquatic organisms even at very low concentrations (Perron et al., 2012). Many other means are also responsible for their toxic accumulation such as their physicochemical metabolism activity as well as photocatalytic activity (Gupta et al., 2015). Due to these processes, these PCPs become non-soluble and non-degradable, but become more prone to react with other PCPs or other contaminants to become more toxic in nature. These personal care products can easily transform into more stable contaminants (Wong-Wah-Chung et al., 2007).

Several research investigations reported that this triclosan is readily mixed with other organic compounds and thus collected in deposits (Fernandes et al., 2011). Some other compounds like phthalate esters, widely used as plasticizers and also in various other consumer goods like detergents and soaps. These are now known to be potential endocrine-disrupting agents for human beings (Gupta et al., 2015). These are biologically degraded by blue green alga by trans-esterification on the adjacent chains of these compounds (Babu & Wu, 2010).

Microalgal populations have been adversely affected by many inorganic and organic substances including personal care products (PCPs, Miazek & Brozek-Pluska, 2019). Pharmaceutical residues and PCPs released from hospitals, industries, houses, and several other facilities can enter ecosystems through wastewaters (Sanderson et al., 2003). Bioremediation of wastewater can be done by living microalgal cells (Coimbra et al., 2018). Microalga has the ability to remove pollutants like pharmaceutical residues and PCPs by bioremediation method (Wang et al., 2017).

7.2.6 *Surfactants*

Surfactants such as nonylphenol ethoxylates, and linear alkylbenzene sulfonate (LAS), are extensively used in domestic as well as industrial detergents. According to studies, it has been estimated that every year, approx. nearly 2 million tons of detergents are manufactured synthetically in the world (de Wolf & Feijtel, 1998). These anionic residual surfactants merge in water bodies with the runoffs (Gupta et al., 2015). These were passes to their water geological cycle and thus their adsorption to the sewage sludge along with the perseverance of detergent molecules, and biochemical structure play a major role in the magnitude of environmental expulsion of detergents and its deprivation by-products (Ojo-Omoniyi, 2013).

In several studies it has been found that bacteria utilized the degraded form of dialkyltetralin sulfonates as its source of sulfur, which is a significant part of commercial LAS and cause of 10–15% impurities (Kölbener et al., 1995; Cook, 1998). Nonylphenol and alkylphenols are broadly used by detergent industry and many other commercial products although they are precursors of alkylphenol polyethoxylates and nonylphenol ethoxylates and well known for their estrogenic effects. Nonylphenol is known as potential endocrine disruptor and xenoestrogen (Maguire, 1999). On studying it has been reported that *Cyclotella caspia* has potential to remove more than 40% of xenoestrogen nonylphenol in 192 hours by the process of biodegradation at the minimal quantity of nonylphenol (Liu et al., 2013), but on increasing concentration of the nonylphenol, the rate of degradation decreases due to its toxic effect (Gupta et al., 2015).

7.2.7 Antibiotics

Antibiotics are substances that act against pathogenic microbes and in livestock for the enhancement of product (Xiong et al., 2021). Recently antibiotic contamination has increased several folds. Low concentration of antibiotic residue can cause adverse effects on organisms directly or indirectly (Huang et al., 2020). Overuse of antibiotics can promote the spread and causes two major concerns regarding human health are: (a) composition of human intestinal microbiome might be altered that lead to the development of antibiotic resistance which will be responsible for the development of severe disease and even death; (b) in the environment they generate environmental antibiotic resistance (Ben et al., 2019). Thus, the environmental spread of antibiotics must be censoriously important to control, especially for WWTPs (Ahmed et al., 2017).

Rather than using older biological method for wastewater treatment, nowadays microalgae-based technology has been promoted more and more because of several advantages such as efficient CO_2 fixation and eco-friendliness (Nguyen et al., 2021). Therefore, it is a promising method for cost-effective remediation of several nutrients and emerging contaminants (ECs) in wastewater (Sutherland & Ralph, 2019).

7.2.8 Persistent Organic Compounds

In the past decades, numerous investigations clearly indicated that these organic compounds are easily degraded and engulfed by microbial inoculants like cyanobacteria and microalgae (Sethunathan et al., 2004). For example, diazinon (insecticide), widely used in agricultural and non-agricultural practices. Astronomically immense amplitudes of these compound residues and their metabolites are being identified underwater during daylight. These include sewage effluents and rural and urban waterways (Moussavi et al., 2013). The presence of diazinon in the microalgal system can deform the chloroplast (Megharaj et al., 1994; Kurade et al., 2016).

Biodegradation and biotransformation of pesticides by microalga must be analyzed for the calculation of risks developed by the by-products on the immediate environment. The remediation of these emerging contaminants is now in practice after introduction of several processes and biological entities. The process of removal of the toxic compound from the environment is considered to be remediation. There are various remediation processes; here we only consider the biological remediation in brief and mainly focus on phyco-remediation (Gupta et al., 2015).

7.3 Bioremediation

Bioremediation or biological remediation is the procedure in which emerging contaminants are degraded using biological inoculants such as microalgae, cyanobacteria, fungi, plants, and several microbes, which makes them lesser toxic or in the form of usable components. These processes involve the uptake of these contaminants by organisms and further catalysis by their enzymes to break down these complex compounds into simpler inorganic forms which are non-toxic to organisms as well as to the ecosystem. Various biological organisms such as plant, fungi, and algae were used in remediation of these emerging contaminants. The present chapter only discussed the role of algae in remediation of these emerging contaminants (Gupta et al., 2015).

7.3.1 Phyco-remediation

The use of microalgae for degradation of these contaminants as well as heavy metals is called phyco-remediation. These algae found well grown in water reservoirs and able to utilize these toxic compounds for their metabolic activities. The use of microalgae for the biodegradability of these emerging contaminants (ECs) makes them a more important bio-tool for remedial applications of these contaminants (Muñoz et al., 2006) and heavy metals (Perales-Vela et al., 2006). These algae have been found growing in wastewater, and various contaminated or stagnant water reservoirs make them more favorite for bioremediation as these are found growing rapidly with higher production of their colonies/hyphae (Olguín, 2012). Recently, several contaminants have emerged and become a major threat to the ecosystem (Prajapati et al., 2013). These include pharmaceutical by-products (drugs, analgesic, antibiotics, etc.), pesticides, etc. These emerging contaminants (ECs) are mixed in natural water reservoirs, as reported by several researchers (Gupta et al., 2015). These chemicals are mixed with other toxic constituents in aquatic resources and thus get adsorbed and immobilized for their further degradation by biochemical process.

There is no clear information about the cost effective, eco-friendly and easy process of bioremediation as compared to physical and chemical treatments (Herbes & Schwall, 1978). But several investigations support the bioremediation especially phyco-remediation for more durable, easy, and high potential in degradation of these contaminants as compared to means of remedial studies (Subashchandrabose et al., 2013). These microalgae and cyanobacteria are found more suitable for removal of these contaminants as compared to other organisms such as bacteria and fungi. These can easily grow in approx. all types of environments and are found flourishing in unfavorable conditions utilizing these compounds (Subashchandrabose et al., 2013). These algal biomasses are found to have potential in degrading wastewater generally nitrogen- and phosphorus-related compounds (Shriwastav et al., 2014).

Several studies proved their ability to degrade the organic compounds such as endocrine-disrupting compounds, and poly-aromatic hydrocarbons. These degradations are done by various physiochemical processes which include bioaccumulation, biodegradation, and bio-adsorption (Matamoros et al., 2015). Various emerging contaminants which are now a major threat to the environment and organism health are discussed below (Gupta et al., 2015).

7.3.2 Limiting Factors of Phyco-remediation

Various factors are accounted for the efficient algal biotransformation and degradation. Synergistic effect of microalgae with surrounding bacterial colonies was significantly investigated to be potentially effective as compared to the single treatment of either (Chekroun et al., 2014). Physiology of algal species is the key factor for phyco-remediation, but some other important factors may also be considered like concentration and physical and chemical properties of effluents (Kobayashi et al., 2006). Degradation of low molecular weight aromatic hydrocarbons maintain the Michaelis-Menten kinetics (Gupta et al., 2015); however, high molecular weight insoluble hydrocarbons such as naphthalene and phenanthrene are non-degradable (Gupta et al., 2015). Many environmental factors like temperature, photoperiod and intensity, pH and redox potential, and salinity directly or indirectly affect the phyco-remediation process (Gupta et al., 2015). All the elements have been recycled by natural biogeochemical cycles (Garrett, 2000). Right from the advent of earth, all elements are present in the natural bodies and biological systems. When the concentration of any particular metal exceeds the tolerance level, then the metal is considered as a contaminant (Dung et al., 2013).

Phyco-remediation is the degradation or detoxifying of harmful constituents using algae. It is proposed by John (2003), as a subcategory of bioremediation.

Different types of mechanism of phyco-remediation could be used based on the nature and composition of wastewater. A separate microalga was used for separate category of wastewater (González-Dávila, 1995). According to some previous studies, removal of antibiotics by microalga include phyco-adsorption, phyco-accumulation, and phyco-degradation (Hena et al., 2021) and can be achieved by the following processes: (a) quick and simple phyco-sorption of pollutants on the cell surface, (b) a sluggish movement of these compounds in molecular or ionic form via algal membranes, and (c) finally, their phyco-accumulation and phyco-degradation takes place (Fig. 7.2; Yu et al., 2017).

7.3.3 Bio-adsorption of Emerging Contaminants (ECs) by Microalgae

Very extensive and clear study of bio-adsorption of ECs by microalgae cells has been done by various workers. Amazingly, dying cells of *Scenedesmus obliquus* and *Chlorella pyrenoidosa* have ability to adsorb approximately 10% of norgestrel and progesterone (Fig. 7.2; Peng et al., 2014).

Cell wall of microalgae is negatively charged due to carboxyl, phosphoryl, etc. Due to extracellular nature of adsorption, the process varies greatly in different chemical compositions (Fig. 7.2). Positively charged species are reactively attracted towards the cell surface of the microalga, resulting ineffective bio-adsorption (Xiong et al., 2018).

7.3.4 Phyco-accumulation of Emerging Contaminants

Phyco-accumulation is an energy-driven uptake of metabolic. Microalgae through biological accumulation are capable of removing or engulfing these emerging contaminants (ECs) as their nutritional food and utilize them to generate the energy for their metabolic process. In some microalgal species like *Chlamydomonas mexicana* and *Scenedesmus obliquus*, carbamazepine has been accumulated (Xiong et al., 2016). Reactive oxygen species (ROS) can be generated in many algal species induced by accumulated ECs and has two definite applications in biological systems; these are as follows: (a) During stabilized conditions, it acts as signaling molecule at different levels of cellular metabolism. (b) But in high aggressive conditions, these ROS lead to cause maximum level of damage to cell and many times led to triggering their programmed cell death mechanism (Fig. 7.2; Kumar et al., 2016). Lethal inhibition expression activity is uncommon (Hlavova et al., 2015). In *Microcystis aeruginosa*, expressing the photosynthetic genes increased after exposure to antibiotics (Liu et al., 2014; Xiong et al., 2018).

7.3.5 Extracellular Degradation of ECs

Microalgae can also evacuate various extracellular polymeric substances (EPS). The EPS can increase the bio-availability of the ECs (Xiao & Zheng, 2016). So the interaction between different ESP and with algal cellular system is capable to induce the extracellular degradation of these contaminants (Fig. 7.2; Flemming & Wingender, 2010). The mechanisms of interaction between these algae with bacterial colonies and their ability of detoxifying these contaminants can provide a better understanding for developing such microbial degrading bio-tool. Due to the interaction of the cell walls of microalgae and these ECs, efficient treatment of wastewater and several other treatment processes of these contaminants can be processed (Fig. 7.2; Fallahi et al., 2021).

To reconnoiter the efficacy of removal of toxicity or neutralizing the sewage water using algal components, the basic necessity was better understanding of their factors which alter or enhance the biological activity of these microalgae to remove contaminants in wastewater. Previously, some attempts are made to detoxify the contaminants in lake and reduce the organic contents in it by the process of phyco-remediation. An experiment was done to check the efficacy of these microalgae and bacterial inoculants consortially as well as singly for removal of these contaminants in activated sludge (Verma et al., 2020). It was very clear from the observations that synergistic efficacy is greater than single inoculants of these biological cultures for wastewater removal (Ahmed et al., 2022).

7.4 New Tactics for Phyco-remediation of Emerging Contaminants

Phyco-remediation by consortia microalgae is the most efficacious bio-tool for releasing electron and energy flow in their mutualistic interaction with bacterial colonies. Conglomerates of microalgae along with soil or water bacteria are capable to detoxify the contaminants and utilize their nutrient from these emerging contaminants with the release of some inorganic compounds as by-products (Xiong et al., 2017b). A novel shortcut nitrogen abstraction process utilizing a phyco-bacterial consortium in photosequencing reaction was also developed (Wang et al., 2015). This consortium inoculant is found to be more efficient in accelerating the degradation process of both types of hydrocarbons, i.e., aliphatic and aromatic from crude oils (Tang et al., 2010).

To obtain the bacterial and microalgal inoculants, two methods, namely, natural as well as artificial substrate colonization, were performed (Liu et al., 2017). The metabolites obtained from these contaminants by process of this consortial inoculants are further used for the growth of algae. Nevertheless, diverse processes incline to be highly stable to their fluctuations with respect to the surrounding conditions and external factors. Due to coalescence of organisms with several types of metabolic activities and their adapting behavioral changes to environmental stress (Wijffels et al., 2013). It is thinkable to amend the microalgal resiliency and their biological degradation ability towards these toxic contaminants after getting adapted with toxic conditions by transmuting their own internal metabolism as well as external conditions (Cho et al., 2016; Aslam et al., 2017). These are the most important aspects to make these organisms (microalgae and cyanobacteria) get acclimated with these emerging contaminants to get better biodegradability activities from these organisms and removal of contaminants from ecological system.

Antecedent investigations found that the adaptation of these microalgae to these toxic stress can make them a better choice or remedial tool (Chen et al., 2015). Co-metabolism of these contaminants is found to be vulnerable towards phycobiont decomposition in a cometabolic process (Peng & Jia, 2013). The removal capability of several antibiotics is done by using the addition of a co-metabolic substrate, acetate, in microalgae (Xiong et al. 2017c). Some of the organic substrates can pose difficulties in the contaminant abstraction (Xiong et al., 2017c). This was found to be obligatory to study the impact of sundry organic toxicants and their coalescences of these abstraction capability of the contaminants when utilized with these phycobionts with enhanced co-metabolic (Xiong et al., 2018).

7.5 Application of Algae in Advanced Oxidation Processes

The advanced oxidation processes are used for the benefit as pretreatment methods, due to which resistant compound can break into simple intermediate biodegradable compounds. These biological processes can easily degrade the contaminants into non-toxic constituents, which also minimal the extra cost while used by chemical and other processes. The wide adaptation of these microalgae to their high level of contaminants makes them suitable for the phycodegradation of organic contaminants and other toxic components under their oxidation process (Osundeko et al., 2014). The natural process of introduction of genetic alteration of photocatalytic gene in these microalgae makes them capable to perform more efficient oxidation processes to catalyze these organic compounds.

The amended efficiency of these naturally altered microalgae found to be more enhanced rate of photosynthesis, peroxidases activity, and several other cellular responses and processes (Zhou et al., 2017). Several studies also provide the direct evidence of their enhanced oxidation processes. Bai et al. (2015) reported that at the molecular level also, there are vast changes to their molecular structure and they become more stable, and it is found that they are up to 50% more efficient in biologically degrading the quinolone by photolysis under the exposure of ultraviolet rays.

7.6 Application of Algae in Phyco-Energy Cells/Batteries

The algal-based energy cells or batteries can be used as a significant method for treatment of emerging contaminants mostly in the case of wastewater and production of electrical energy and many other valuable products from it (Li et al., 2014). With the help of MFCs, organic carbon can be easily converted into electricity. This technology is not much cost-effective (Li et al., 2014). Algal-based energy cells have minute capacity for removal of phosphorus (Zhang et al., 2011). The arrival of new microalgae techniques is now a good bio-tool for making algal-based energy cells. Symbiotic relationship of these alga with their neighboring bacterial colonies can be a new and valuable approach for energy production and removal of organic contamination (Li et al., 2014). The consortia of MFCs and microalgae continuously produced energy without any external input source under illuminating condition (He et al., 2009). The above results are sufficient for clarifying the integrated role of MFCs and microalgae is a breakthrough in wastewater management (Xiong et al., 2018).

7.7 Application of Genetically Modified Algae in Remediation Potential

The major challenge to increase the phyco-remediation is the population growth of microalgae. The slow growth of these algae creates the problem in increasing of remediation process. The genetic engineering technology here plays the vital role in creating genetically modified microalgae which are able to grow much faster and also perform enhanced remediation processes of these emerging contaminants. These GM microalgae can be beneficial in industrial application and provide the modified metabolic pathways. These are able to provide high product yield and design and construct the artificial photo-autotrophs, which are rising and much promising bio-tools in the coming future (Xiong et al., 2018).

Several biologists believe in this approach and are working to develop the novel microalgae which is genetically modified and can perform higher remedial processes. There are several reports regarding the making of laccase enzyme-based genetically modified microalgae for the removal of these contaminants (Xiong et al., 2018). Laccases belong to the oxidoreductase class of enzymes, and these are known to be bio-catalysts. This is because of their utilization of atmospheric oxygen as electron acceptor and release of water as by-product after their metabolic activities (Xiong et al., 2018).

The genetically modified laccase enzyme activity was observed in several organisms such as *Streptomyces coelicolor* (Toscano et al., 2013), green microalgae such as *Tetracystis* species, *Chlamydomonas* species (Otto et al., 2015), and *Spirulina platensis* (De-Bashan & Bashan, 2010). Several other microalgae and cyanobacteria have also been used with introduced laccase activities and are found more suitable for remediation of these emerging contaminants. These enzymatic actions are more active in alkaline medium/environment. These well degraded the endocrine disruptors and phenolic contaminants (Afreen et al., 2017).

Conclusion

The increase in population also increases the use of products by consumers, and several emerging contaminants comes with the biggest challenge and major concern due to their resistance to degradation as well as their high water repelling nature. These are emerged out to cause several diseases and increases the toxicity in all types of ecosystem. These emerging contaminants are becoming major concern for their proper removal. Several new techniques are introduced by researchers to remediate these contaminants in which the most reliable biological organisms were microalgae and cyanobacteria. The process of removal of these contaminants is called phyco-remediation. The emerging contaminants are endocrine-disrupting compounds, pharmaceutical compounds, antiepileptic drugs, antibiotics, dyes, wastewater contaminants, organic and household wastes, personal care products, etc. The new technologies like genetic engineering incorporated with these microalgae are a promising tool to cope up with these emerging contaminants and help in detoxification of these contaminants from the ecosystem.

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Chapter 8
Contamination of Sewage Water with Active Pharmaceutical Ingredients: An Emerging Threat to Food Products and Human Health

Sadia Sultana, Muhammad Sabir, Saif Ullah, Hamaad Raza Ahmad, and Ghulam Murtaza

Abstract The rapid population expansion in recent years, level of urbanization, and accessibility to healthcare are the main causes of rising active pharmaceutical ingredient (API) emission in the environment. The primary drivers of rising pharmaceutical production—poor laws, self-medication, the dispensing of antibiotics without a prescription, and the use of illicit drugs—lead to their appearance as quickly growing natural pollutants. For the past three decades, pharmaceutical residues have been discovered in nearly every natural setting on every continent, including groundwater, wastewater treatment facilities, effluent, and influent as well as surface water (lakes, rivers, streams, estuaries, and ocean). According to reports, pharmaceutical contamination is currently present in the tropical regions and adjacent areas of the world. Sludge and wastewater from municipal wastewater treatment plants, as well as businesses, hospitals, and homes, are the primary sources of APIs. Significant amounts of pharmaceuticals are released and accumulate in the environment as a result of wastewater application for irrigation and biosolids used as organic fertilizers. The fact that APIs tend to be less volatile, highly polar, and hydrophilic makes them "compounds of rising concern." APIs can become inactive by soil contact, sorption, discharge, or environmental or microbiological change. The potentially dangerous consequences of these active compounds on the ecosystem have just recently come to light because they constitute a life-threatening hazard to soil biota, aquaculture, plants, animals, and ultimately humans. There must be a variety of legal, therapeutic, and disposal solutions that are not only effective but

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S. Sultana (✉)
Institute of Soil and Environmental Sciences, University of Agriculture, Faisalabad, Pakistan

Institute of Soil Chemistry and Environmental Sciences, Ayub Agricultural Research Institute, Faisalabad, Pakistan

M. Sabir (✉) · S. Ullah · H. R. Ahmad · G. Murtaza
Institute of Soil and Environmental Sciences, University of Agriculture, Faisalabad, Pakistan

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also technically and financially feasible because there are so many different types of APIs. The entry of APIs into the food chain must also be carefully investigated in order to develop methods to decrease their impact on biodiversity.

Keywords Active Pharmaceutical Ingredients · Occurrence · Types · Fate · Remediation

8.1 Introduction

The substances that are capable of causing certain conditions or treating the body are known as active pharmaceutical ingredients, or APIs. Wastewater, biosolids, soils fertilized by biosolids, irrigation runoff from fields, and groundwater systems that receive treated/clean wastewater have all been reported to contain these active components (Miller et al., 2016). The usage of biosolids on land and the use of polluted water can transport the APIs, which are water soluble by nature, to the earth's surface and the water, potentially causing a buildup in plants (Wu et al., 2010). These exhibit detrimental effects on organisms, are resistant to degradation, are persistent in aquatic systems, and also have a negative impact on human health (Xu et al., 2009). Numerous investigations have verified the existence of different pharmaceuticals in varied environments. APIs are regarded as an exclusive group of developing environmental pollutants, raising concerns about the potential negative effects on people and wildlife due to their inherent capacity to cause biological consequences in humans at low concentrations. Antibiotics and illegal drugs are just two examples of the large variety of pharmaceutically active products that are produced on a massive scale globally.

The legal requirements for buying and selling drugs vary between nations. Self-medication is popular in poor nations, where it is simple to buy medications from drug stores without a prescription; however, the laws forbid the selling of nonprescription antibiotics (Al-Azzam et al., 2007). However more than 60% of pharmacists give over-the-counter antibiotics for conditions including a sore throat, urinary tract infection, diarrhea, etc. (Almaaytah et al., 2015). Although data on drug use is present in many countries, it is not enough to create a national drug use profile (Beek et al., 2016). On the other hand, medicines are also offered for sale as unlicensed over-the-counter drugs in many nations. As a result, this yard sale causes more harm to consumption estimates and settings (Kümmerer, 2009b). According to statistics, North America, which accounts for 46% of the worldwide pharmaceuticals market in 2020, is followed by Asia Pacific (26%), with Africa having the smallest share (Research and Markets).

The persistence of persistent organic pollutants (POPs) or bioaccumulative compounds and toxins (PBT) in the environment is a crucial indicator for their detection. Under many regulatory systems, the ability of chemicals to persist in the environment is specified by the benchmark set based on their transformation half-lives in the specific medium (air, water, soil, and sediment). Both the Canadian

Environmental Protection Agency and the United Nations Environmental Program have set a half-life of 60 days in water and 180 days in soil as a requirement for chemical persistence (UNEP, 2001; Hughes et al., 2009). The persistence and conversion of half-life of a chemical in a mass are measured using a variety of processes, including biodegradation, hydrolysis, redox reactions, and direct or indirect photolysis.

Redox state, temperature, salinity, microbe activity, and sun exposure are only a few of the environmental elements that have a big impact on these processes (Schwarzenbach et al., 2003). Therefore, a chemical's half-life in the environment can be affected by both chemical-specific properties and ambient factors, and this is why the transformation of pharmaceuticals is primarily dependent on environmental factors (Zou et al., 2014). While medications have been examined in a variety of environmental scenarios over the past two decades (Bu et al., 2013; Richardson et al., 2005), little is known about how they affect the environment (e.g., persistence).

Medicine cannot be categorized as a collection of the same physical, chemical, structural, or biological properties as opposed to homogenous molecules like chlorofluorocarbons (CFCs), polychlorinated biphenyles (PCBs), or polycyclic aromatic hydrocarbons (PAHs). They might have polymorphism, many ionization sites, and complex chemical structures (Taylor & Senac, 2014). In medicine, polarity is typically spontaneously created for a particular purpose in a target object. Medical pollution differs from many other contaminants based on: (i) having molecular masses <500 Dalton (allow skin absorption), some have large molecular masses (Lipinski et al., 1997); (ii) comprising chemically complex molecules (variety of structures, shape, molecular mass, and function); (iii) involving a polar compound with more than one detectable group. In pharmaceuticals, some structures (iv) are dependent on the pH of the prevailing medium, (v) they are lipophilic or hydrophilic. As a result, they can (vi) endure in the natural world, multiply throughout life, and continue to be physiologically active (Rivera-Utrilla et al., 2013). For instance, whereas clofibric acid stays unaltered for many years, naproxen, sulfamethoxazole, and erythromycin may linger in the area for up to a year. Due to their propensity to (vii) adsorb and spread throughout a live organism, these molecules' chemical structures change metabolically (Ribeiro et al., 2012).

8.1.1 Emergence of Active Pharmaceutical Ingredients in Contaminated Water and Solid Waste

Active pharmaceutical ingredients (APIs) have recently been discovered as micropollutants in a multitude of environments, including drinking water (Stackelberg et al., 2004), agricultural soils, groundwater (Lindsey et al., 2001), and surface water (Kolpin et al., 2002). Pharmaceuticals are typically discovered in the aquatic environment at very high elevations, which may be because they are continuously excreted from the sewage system at rates that are much faster than their natural excretion rates (Daughton, 2003). Others, such acetylsalicylic acid (ASA) and diclofenac acid, are detected in several water bodies in Spain, Italy, Germany, Canada, Brazil, Greece, and France at amounts of 0.22 and 3.02 mg/L, respectively. Each year, approximately 500 tonnes of analgesics are reportedly produced locally (Heberer, 2002). An American case study revealed that solid municipal trash collected at the MSW transmission station in Orange County, Florida, had 8.1 mg/kg quantities of 22 APIs, including ciprofloxacin, acetaminophen, ibuprofen, trimethoprim, nicotine, and clotrimazole (Musson & Townsend, 2009). Ibuprofen (77–202 g/L), salicylic acid (5.3–150 g/L), clofibric acid (10.4–158 g/L), and other APIs were found in significant amounts in leachates starting at midnight, according to a study done in Guangzhou waste disposal sites (Peng et al., 2014). The amount of API residues, such as antibiotic residues, has been measured at comparatively high amounts in developing countries, compared to industrialized countries, which lack advanced infrastructure for the treatment of waste water (Zhang et al., 2012). Ibuprofen (1673 mg/L, 6046 mg/kg, 1229 mg/kg, and 610 mg/kg), diclofenac (836 mg/L, 4968 mg/kg, 6632 mg/kg, and 257 mg/kg), and naproxen (464 mg/L, 7273 mg/kg, 4819 mg/kg, and 199 mg/kg) all had high concentrations of APIs in waste water, sludge, solid waste, and soil samples (Ashfaq et al., 2016).

Sewage Water

Pharmaceuticals have been discovered in the larger watershed, and it has been established that they could be harmful to people and other animals, watery habitats' inhabitants, and biota. As a result, there is increased interest in the toxicity of APIs for humans and the environment (Boxall et al., 2012). Although some of these substances are nonbiodegradable and resistant to transformation, household chemicals and agricultural drugs were previously not thought to be environmental pollutants. As a result, these substances have built up in water bodies. Understanding their emission routes is crucial to reduce pharmaceutical pollution in the environment (Qingwei et al., 2016; Bound & Voulvoulis, 2005). Active pharmaceutical chemicals have also been discovered to reach the aquatic environment through manufacturing enterprises, hospitals, residential, agricultural, and industrial usage. These medications either represent manufacturing waste, consumer waste, or improperly disposed of expired ones.

The degradation process starts once the APIs are in place, and a number of intermediary molecules cause issues when metabolites and parent chemicals are naturally analysed. Due to the fact that many APIs are resistant to removal in sewage treatment plants, it has been determined that both untreated and treated sewage water are plentiful in APIs (Gros et al., 2010; Hughes et al., 2013). According to several investigations, active medicines can be found in both treated wastewater and soil that has been fertilized with biosolids or irrigated with it (Kinney et al., 2006). The frightening fact regarding pharmacological events and outcomes is that living things are exposed to them through a wider number of routes than is recommended on a daily basis (Kümmerer, 2008a, 2008b). The continued release of pesticides into the environment in a variety of ways can result in serious health hazards for aquatic life, plants, and in even low amounts. Consequently, it is imperative to conduct drug detection tests in water (Wu et al., 2014).

Solid Waste

Pharmaceuticals can enter the aquatic environment through domestic dumping (Bound & Voulvoulis, 2005). Similar to this, dumping of unwanted pharmaceuticals in household solid wastes poses a serious threat to the environment (Massoud et al., 2016). As a result, pharmaceutical pollution in the environment may be significantly influenced by non-wastewater emission pathways.

8.1.1.1 Types of Different Active Pharmaceutical Ingredients

Modern medicine plays a crucial role in daily life and has a positive impact on society. Prescription drugs for both humans and animals employ more than 4000 active pharmacological components worldwide. Active ingredients are made up of a variety of synthetic chemical compounds and are produced by pharmaceutical businesses in the region of 100,000 tonnes annually (Weber et al., 2014). Concern has been raised over the past two decades about the intentional presence of APIs in quantities that could harm aquatic creatures in a variety of aquatic habitats (water, sediments, and biota). The term “active pharmaceutical ingredients” (APIs) refers to a broad category of organic chemicals, including pharmaceutical, phytotherapeutic, biotechnological, veterinary, fragrance, and cosmetic items. Often, these components’ solubility and a range of physicochemical parameters, such as the partition coefficient octanol-water (Kow), the coefficient of distribution-biosolids-water (Kp), and others, explain environmental dynamics (Kümmerer, 2008a, 2008b) (Table 8.1).

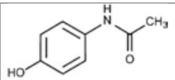
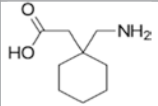
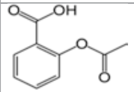
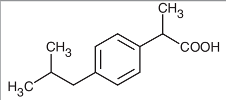
Analgesics

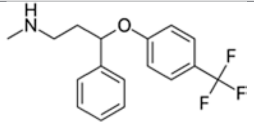
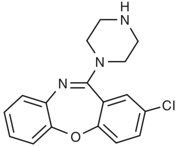
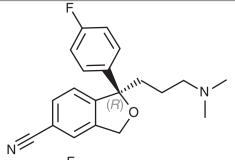
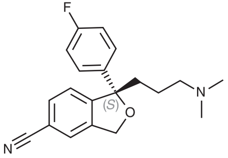
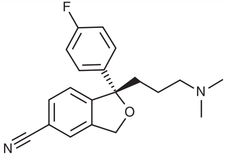
Analgesics, which are self-medication drugs that are often used worldwide, are a growing public health concern (Ministerio et al., 2004; Van Gerven et al., 1996). Surface-water studies reveal the presence of pharmaceutical ingredients such ASA, naproxen, ibuprofen, diclofenac, and ketotifen as well as other ibuprofen degradation products like hydroxy-ibuprofen, carboxy-ibuprofen, and carboxyhydrotropic acid, which are more hazardous than their constituents (Buser et al., 1999).

Antihypertensive

Globally, malignant hypertension is the most frequent cause of heart disease. Angiotensin converting enzyme (ACE) inhibitors, beta blockers, and calcium channel blockers, which are commonly used hypertension medications, have been found in water in recent years (He & Whelton, 1997). Atenolol, metoprolol, and propranolol are a few beta blockers that lower blood pressure. Their levels in sewage from municipal wastewater have increased to more than 0.017 g/L, and they negatively affect aquatic life (Ternes, 1998).

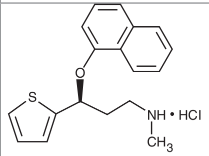
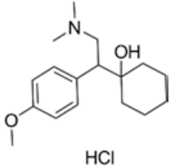
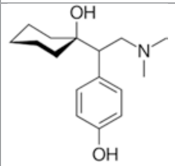
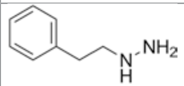
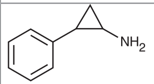
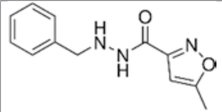
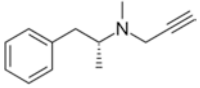
Table 8.1 Properties of Active Pharmaceutical ingredients

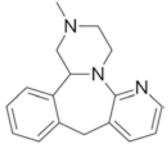
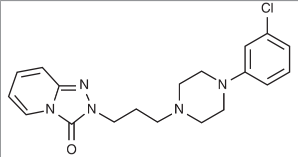
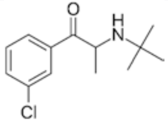
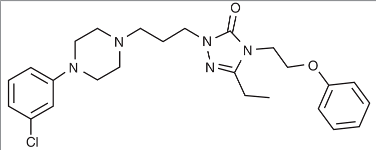
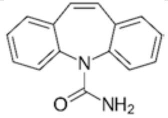
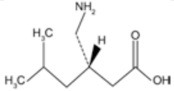
Group	Emerging Contaminants		Formula	Structure	Log K_{ow}	pKa	½ life
Analgesics							
Non-opioid analgesics	Acetaminophen	Paracetamol	$C_8H_9NO_2$		0.46	9.38	2 hrs
	Anticonvulsants	Gabapentin	$C_9H_{17}NO_2$		−1.10	3.6	5–7 hrs
	Aspirin	Acetylsalicylic acid	$C_9H_8O_4$		1.19	3.47	15–20 min
	Other NSAIDs	Ibuprofen	$C_{13}H_{18}O_2$		–	5.3	3.1–7 days
Adjuvant analgesics	*Antidepressants*						

	Tricyclic antidepressants (TCA)	Fluoxetine	$C_{17}H_{18}F_3NO$		–	9.8	1–4 days
		Amoxapine	$C_{17}H_{16}ClN_3O$		–	8.71	8 hrs
	Selective Serotonin Reuptake Inhibitors	Citalopram (Celexa)	$C_{20}H_{21}FN_2O$	 	1.39	9.78	35 hrs
		Escitalopram (Lexapro)	$C_{20}H_{21}FN_2O$		3.74	9.80	27–33 hrs

(continued)

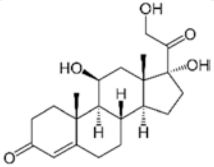
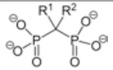
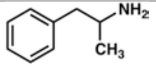
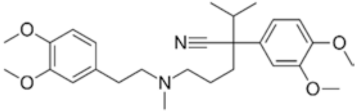
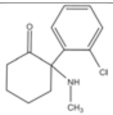
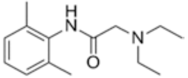
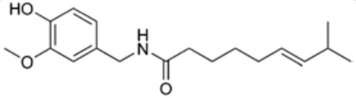
Table 8.1 (continued)

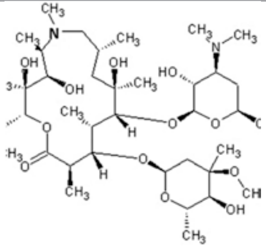
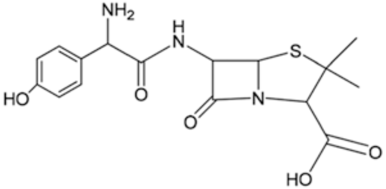
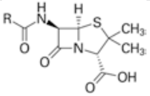
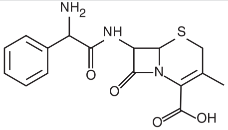
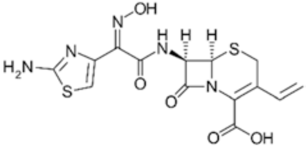
Group	Emerging Contaminants		Formula	Structure	Log K_{ow}	pKa	½ life
	Serotonin Norepinephrine Reuptake Inhibitors	Duloxetine (Cymbalta)	$C_{18}H_{19}NOS{\bullet}HCl$		4.68	9.70	8–17 hrs
		Venlafaxine (Effexor)	$C_{17}H_{27}NO_2$		3.20	10.09	5 hrs
		Desvenlafaxine (Pristiq)	$C_{16}H_{25}NO_2$		2.72	9.45 (amine) 10.66 (phenol)	11 hrs
	Monoamine Oxidase Inhibitors	Phenelzine (Nardil)	$C_8H_{12}N_2$		0.97	6.5–8	11.6 hrs
		Tranylcypromine (Parnate)	$C_9H_{11}N$		1.58	9.68	2.5 hrs
		Isocarboxazid (Marplan)	$C_{12}H_{13}N_3O_2$		–	10.4	1.5–4 hrs
		Selegiline (EMSAM, Eldepryl)	$C_{13}H_{17}N$		–	7.48	1.5 hrs

	Other Antidepressants	Mirtazapine (Remeron)	$C_{17}H_{19}N_3$		–	7.7	20–40 hrs
		Desyrel (trazodone)	$C_{19}H_{22}ClN_5O$		3.21	6.79	10–12 hrs
		Bupropion (Wellbutrin)	$C_{13}H_{18}ClNO$		3.85	8.22	21 hrs
		Nefazodone (Serzone)	$C_{25}H_{32}ClN_5O_2$		5.0	6.5	5 hrs
	Anticonvulsants	Carbamazepine	$C_{15}H_{12}N_2O$		1.51	13.9	35 hrs
		Pregabalin	$C_8H_{17}NO_2$		1.78	10.23	6.3 hrs

(continued)

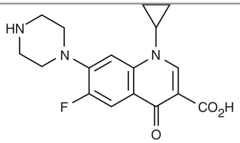
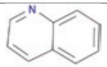
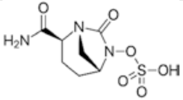
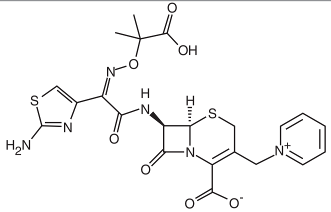
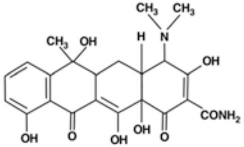
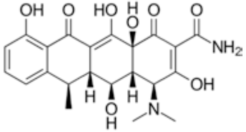
Table 8.1 (continued)

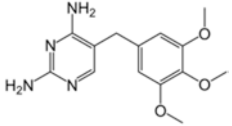
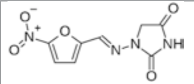
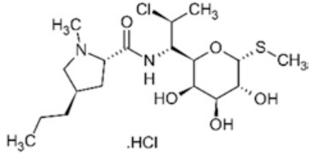
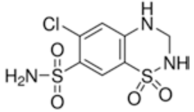
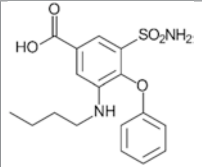
Group	Emerging Contaminants		Formula	Structure	Log K_{ow}	pKa	½ life
	Anti-inflammatory	Corticosteroids	$C_{21}H_{28}O_5$		1.62	12.59	8–72 hrs
	Bisphosphonates		$C_4H_{13}NO_7P_2$		−0.5	2.72	0.5–2 hrs
	Stimulant	Amphetamine	$C_9H_{13}N$		1.76	10.13	10 hrs
	Calcium channel blockers	Verapamil	$C_{27}H_{38}N_2O_4$		3.79 at pH 9.0 2.15 at pH 7.0	8.92	2.8–7.4 hrs
	Anesthesia	Ketamine	$C_{13}H_{16}ClNO$		2.18	7.5	10–15 min
		Lidocaine	$C_{14}H_{22}N_2O$		2.26	7.8	1.5–2 hrs
	Sodium channel blockers	Capsaicin	$C_{18}H_{27}NO_3$		3.04	9.93	1.64 hrs
	Radioisotopes	Strontium 89	$SrCl_2$	$Cl^{-}\,Sr^{2+}\,Cl^{-}$	–	–	50.5 days
	Antibiotics						

Macrolides	Azithromycin	$C_{38}H_{72}N_2O_{12}$		8.74	8.5	68 hrs
Penicillin	Amoxicillin	$C_{16}H_{19}N_3O_5S$		0.87	2.6	60 min
	Penicillin	$C_{16}H_{18}N_2O_4S$		1.83	2.6–2.8	1.4 hrs
Cephalosporins	Cephalexin	$C_{16}H_{17}N_3O_4S$		–	5.2–7.3	60 min
	Cefdinir	$C_{14}H_{13}N_5O_5S_2$		–	3.35–9.7	1.7 hrs

(continued)

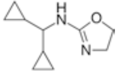
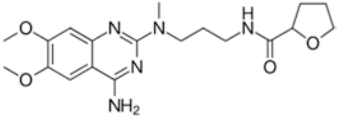
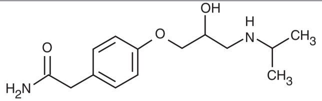
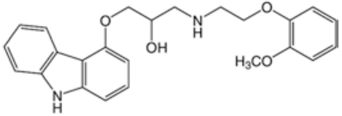
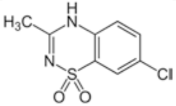
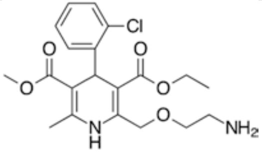
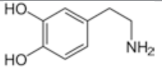
Table 8.1 (continued)

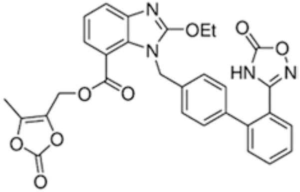
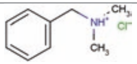
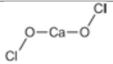
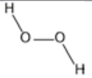
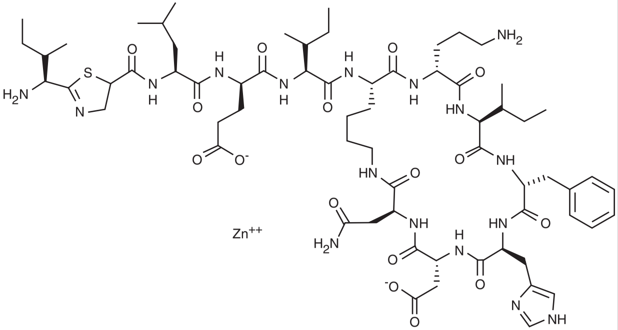
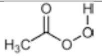
Group	Emerging Contaminants	Formula	Structure	Log K_{ow}	pKa	½ life
Fluoroquinolones	Ciprofloxacin	$C_{17}H_{19}ClFN_3O_3$			6–8.8	4 hrs
	Quinolone	C_9H_7N		2.03	4.90	1.5–16 hrs
Beta-lactams	Avibactam	$C_7H_{11}N_3O_6S$		–	−3.9	2 hrs
	Ceftazidime	$C_{22}H_{22}N_6O_7S_2$		–	2.77–4.26	1.9 hrs
Tetracyclines	Tetracyclines	$C_{22}H_{24}N_2O_8$		−1.37		57–108 hrs
	Doxycycline	$C_{22}H_{24}N_2O_8$		–	3.09	18–19 hrs

Trimethoprim-sulfamethoxazole	Trimethoprim	$C_{14}H_{18}N_4O_3$		0.91	7.12	8–10 hrs
Urinary anti infection	Nitrofurantoin	$C_8H_6N_4O_5$		−0.47	7.2	30 min
Lincosamides	Clindamycin	$C_{18}H_{34}Cl_2N_2O_5S$		–	7.56	2.4 hrs
Antihypertensive						
Diuretics	Hydrochlorothiazide	$C_7H_8ClN_3O_4S_2$		−0.07	7.9	8–12 hrs
	Bumetanide	$C_{17}H_{20}N_2O_5S$		1.67	3.83	0.75–2.6 hrs

(continued)

Table 8.1 (continued)

Group	Emerging Contaminants		Formula	Structure	Log K_{ow}	pKa	½ life
Sympatholytic Drugs	Centrally acting antiadrenergic agents	Rilmenidine	$C_{10}H_{16}N_2O$		–	–	8 hrs
	Alpha-Adrenergic Blocking Agents	Alfuzosin	$C_{19}H_{27}N_5O_4$		1.86	8.13	3–5 hrs
	Beta-Adrenergic Blocking Agents	Atenolol	$C_{14}H_{22}N_2O_3$		0.16	9.58	6 hrs
	Alpha- Beta-Adrenergic Blocking Agents	Carvedilol	$C_{24}H_{26}N_2O_4$		4.19	7.97	6 hrs
Vasodilators	Potassium channel openers	Diazoxide	$C_8H_7ClN_2O_2S$		1.67	8.74	5.9–27.7 hrs
	Calcium Channel Blockers	Amlodipine	$C_{20}H_{25}ClN_2O_5$		3.0	9.26	35–50 hrs
	D1-dopamine receptor agonists	Dopamine	$C_8H_{11}NO_2$		−0.98	8.93	80.2 hrs

Angiotensin inhibitors and antagonists	Angiotensin II Receptor Blockers (ARBs)	Azilsartan	$C_{25}H_{20}N_4O_5$		6.44	6.1	11 hrs
Sanitizing Agents							
Quaternary Ammonium	Alkyl dimethyl benzyl ammonium chloride		$C_{22}H_{42}ClNO$		3.91	–	7.1 days
Hypochlorite	Calcium Hypochlorite		$CaCl_2O_2$		–	–	–
Accelerated Hydrogen Peroxide			H_2O_2		–	–	30 days
Phenolics	Bacitracin		$C_{66}H_{103}N_{17}O_{16}S$		–	3.19	2–4 days
Peracetic Acid			$C_2H_4O_3$		−1.07	8.20	22 min

Antibiotics

Drugs that fight bacteria in both people and animals are frequently prescribed antibiotics. However, their overuse has reduced the effectiveness of treatments and promoted microbial resistance (Kumar et al., 2005; Wright, 2005; Lambert, 2005). Additionally, some infections' resistance mechanisms have been boosted by the presence of antibiotic residues in various water bodies. Vancomycin, beta-lactams, macrolides, aminoglycosides, tetracyclines, and other antibiotics have all been discovered in the water (Roberts et al., 1999; Akinbowale et al., 2007). The parent components or products generated are collected from hospitals, homes, or agricultural origins and then disposed of directly in contaminated water. Salmonella, Pseudomonas, Escherichia coli, Aeromonas, and Staphylococcus are only a few of the pathogens that have benefited from the high concentration of antibiotics in the water (Baquero et al., 2008). Due to the inefficiency of antimicrobials used to treat illnesses, if these antimicrobial viruses infect individuals directly or through carriers, they may increase the number of deaths among hosts. Ironically, the identification of waterborne pathogens is a significant problem that needs to be addressed in legal precedents, regulated, and monitored for the presence of antimicrobials in contaminated water, as well as for proper disposal, in order to prevent public health issues across the globe.

Sanitizing Agents

Antimicrobials triclocarban and triclosan (sanitizing agents) are found in high concentrations of all compounds measuring 2715 and 1265 μg kg^{-1} dry weight or 52% and 24%, respectively, of all existing APIs mass (Evelyn et al., 2010). This high concentration may be due to its overuse and the strong tendency of the two antimicrobials to separate and persist in sewage sludge. By comparison, triclocarban and triclosan released from the US sewage system is >330,000 and >300,000 kg yr^{-1}, respectively (Halden & Paull, 2005).

8.1.1.2 Sources of Production of Active Pharmaceutical Ingredients

Active pharmaceutical ingredients enter the environment through different routes. Largely, there are five major sources responsible for release of pharmaceutical drugs and their intermediate compounds into the environment. The major sources of APIs are industries, hospitals, households, wastewater, and sludge from municipal wastewater treatment plants from where these ingredients are released into the environment due to inappropriate management of drug-containing waste. Application of wastewater for irrigation and biosolids as organic fertilizers are responsible for release and buildup of substantial amounts of drugs in the environment, e.g., fluoroquinolones, ciprofloxacin, and norfloxacin. These pharmaceutical compounds, which are resistant to degrade in the wastewater treatment practice, gather in substantial magnitudes in solid waste and ultimately in soil (Hamscher et al., 2001; Kuhne et al., 2001). A non-point source of contamination is soil fertilization with animal excreta. According to an estimation, veterinary pharmaceutical ingredients excretions are many folds higher than that of human drug application. That is why, veterinary medicines accumulate in soil and seep into the underground water resources more than surface water resources (Sarmah et al., 2006). These veterinary medicines and their degraded compounds contaminate the soil and find their way into the food web. Freshwater is also contaminated by APIs through runoff from agricultural land fertilized with processed solid waste (Nikolaou et al., 2007).

(a) *Hospitals*

The quantity of APIs in wastewater excreted from hospital is greater than in civic waste (Schuster et al., 2008). Conversely, the entire chemical drift is lesser because in developed countries, hospital effluent gets diluted by municipal waste (Kümmerer & Helmers, 2000).

(b) *Households*

Over-dated medicines or their residuals from households are generally dumped into the sewerage system. Patients tend to keep medicines at home because of change in prescription or self-discontinuation (Abahussain et al., 2006). Approximately one-third of pharmaceuticals sold is either flushed (11.2%) or thrown into the bin (76.5%) (Rönnefahrt, 2005). Only about 22.9% patients return medicines to a drug store for dumping (Seehusen & Edwards, 2006). Injudicious discarding of unused medicines should be addressed wisely (Bound & Voulvoulis, 2005). Hence, patient should be educated about expired and unused medicine's disposal.

(c) *Landfills*

The final destination of solid waste is landfill. After being thrown off into public solid waste, medicines in a landfill may experience breakdown, adsorption, or leach down and eventually leave the dumping site (Musson & Townsend, 2009). So, if landfill sites are not insulated then they might be a big source of groundwater pollution. Organic waste from the pharmaceutical production units moves down the gradient of a dumping site/landfill (Eco-Cycle, 2011). Some medical substances (different sulfonamides and propylphenazone) are found in the adjacent aquifers as a part of the leachates being used during 1940s–1970s for human medication (Holm et al., 1995).

8.1.2 Factors Affecting Active Pharmaceutical Ingredients

Active substances enter land and aquatic environments depending on factors like market demand, recurrence rate, self-medication, and drug use. Certain elements, such as demography, economy, illness loads, governmental legislation, and cultural preferences, have an impact on the overall usage and consumption of APIs within a nation.

(a) *Population*

The proportion of people in a country and their access to healthcare are two important factors that affect the overall demand for APIs. A good predictor of API environmental discharge is the level of urbanization (Verlicchi et al., 2012). However, there are other factors that can reduce the environmental danger; therefore, population pressure (or lack thereof) does not simply translate into environmental damage. The demand for medications is significantly influenced by the age demography. As a result, as people age, there is a greater per capita need for medication. However, the overall use of APIs is a complex phenomenon; for the older population, various pharmacological combinations are anticipated than for a younger population, not only in terms of quantity but also in terms of importance (Kookana et al., 2014). It is interesting to note that low-income continents have fewer elderly people than high-income continents; for example, just 7.1% of the total population in Asia is currently over 65, compared to 13.4% in North America and 16.4% in Europe (Population Reference Bureau, 2013). Younger or older populations may utilize APIs in various ways and to varying degrees, which could affect their potential ecological footprints.

(b) *Usage Trends*

The volume and pattern of an API's usage are the most significant factors influencing the ecological risk of that API. Monitoring the levels of API in sewer systems and "back-calculating" the average consumption in that region can be used to estimate provincial API use (Lai, 2011). The sale of APIs can also be used to estimate API usage, however this information is highly reliant on other factors and is therefore difficult to translate into an estimate of the volume of APIs utilized (Kostich et al., 2010). Although it varies by area and is approximately 50% in high-income countries and even lower in lower-income economies, long-term treatment measurement is still a significant role (Sabate, 2003).

(c) *Self-medication*

The selection and use of APIs by individuals to address self-recognized ailments or symptoms is what WHO defines as "self-medication" (WHO, 1998). This widespread practice is especially prevalent in low-income countries (Asia, Africa, and Latin America) with lax medication sales regulations. Antibiotic overuse causes resistance to develop, lateral effects, allergic reactions, and poisoning (Kamat & Nichter, 1998). Furthermore, in lower-income nations, overprescription or prescription based on perceived patient demand is an issue. Other factors that may affect a doctor's prescribing choices include vested interests (affiliations with a pharmacy, brand, or production facility) and any incentives provided by the pharmaceutical business (Radyowijati & Haak, 2003).

(d) *Chronic Diseases Burdens*

APIs are overused as a result of sophisticated urbanization, lifestyle changes, and increasing per capita income. Increased affluence affects the pattern of illness burden by favoring chronic or noncommunicable diseases in comparison to acute or infectious diseases, thanks to urbanization and income. Because some chronic diseases, including diabetes, are increasingly prevalent in some lower-income nations, the use of APIs has increased (Hoebert et al., 2011). Additionally, in Asian, African, Latin American, and Pacific countries, mass drug administration (MDA) initiatives are being employed to combat neglected tropical diseases such as lymphatic filariasis and leprosy (Liese et al., 2010). As a result, there are more APIs released into the environment during certain seasons (WHO, 2013), which increases awareness and impact of these programs.

(e) *Veterinary Drugs*

Significant volumes of veterinary antibiotics (antimicrobials and parasiticides) are used in lower-income nations, notably those in Asia, due to the high densities of the livestock and aquaculture industries (Heberer, 2009; Rico & Van den Brink, 2014; FAO, 2007, 2012). The emergence of microbial resistance is a possible danger connected to the use of antibiotics in aquaculture (Rico & Van den Brink, 2014).

8.2 The Fate of Active Pharmaceutical Ingredients

API distribution via the environment primarily occurs by aqueous transport and food chain dispersal since these are mostly less volatile, highly polar, and hydrophilic (Caliman & Gavrilescu, 2009). Once released into the environment, depending on the properties of the receiving environment and the molecule, APIs may be *transported* over vast distances. Depending on the characteristics of the particular molecule, pharmaceuticals undergo a variety of processes in the aquatic environment, including aerobic and anaerobic biodegradation, abiotic transformation, hydrolysis, and sediment sorption. When sludge is put to agricultural land as fertilizer or to areas that get treated wastewater irrigation, certain groups of APIs present in the sludge can find their way into the environment through adsorption (Pedersen et al., 2005; Van Wieren et al., 2012). Resultantly, these active pharmaceutical ingredients are taken up by the crops, which may then enter the human system (Wu et al., 2014, 2015).

API *runoff* from agricultural land or landfills containing biosolids has the potential to contaminate nearby surface waters or seep into the groundwater (Kleywegt et al., 2007) thereby posing a risk to aquatic life and public health. *Sorption* to the sediment is another method by which APIs are transferred to the aquatic environment. These active pharmaceutical pollutants are accumulated in the sediment, which serves as a sink and may later release them back into the aquatic environment (Zhao et al., 2013). Additionally, some APIs are more persistent in sediment than in water, including sulfamethoxazole, carbamazepine, triclosan, and ciprofloxacin (Chenxi et al., 2008; Conkle et al., 2012). However, the bioavailability or toxicity of APIs may not necessarily be decreased by their adsorption to sediment or suspended particles, which may affect their active concentrations in the aquatic environment.

Continual exposure to these substances in the sediments may also have negative impacts on benthic creatures (Gilroy et al., 2012).

Persistence
As pharmaceuticals are a necessary evil for human benefits, the consumption and consequent release to the environment are inevitable. Hence, the presence and the persistence of pharmaceuticals in environments are of great concern to understand their environmental fate. Through prolonged observation or biomonitoring, it is possible to infer persistence from a compound's ongoing presence in an environment removed from the emission source. However, as medications usually have significant residual concentrations in the aquatic environment and a sluggish rate of change, this is not really applicable to the drugs in habitats (Daughton 2005). Another striking challenge of APIs examination is the variations in half-lives obtained in the laboratory and the natural environment, e.g., carbamazepine's half-life exceeds 60 days in laboratory conditions in aqueous media, although the long half-life of 1200 days is also observed in field conditions (Zou et al., 2015). The half-life of Diclofenac in the natural environment is also shorter than 60 days, according to research, albeit the measured values of half-life vary between studies. Ibuprofen has been discovered to have a photolysis half-life of 413 days, and a half-life that is less than the persistence threshold (60 days) is also recorded (Yamamoto et al., 2009).

The most crucial pace is the development of advanced tools for accurately estimating drugs' persistence in the field and laboratory. For determining persistence in the field (river or lake), the mass balance model is a well-defined and significant method (Fono et al., 2006). Although chemical fluxes, particularly for pharmaceuticals in the real environment, are generally time-dependent and also vary with location, mass balance calculations assume a steady state (Ort et al., 2014). Therefore, the ongoing presence of medicines in the aquatic environment presents a significant challenge to the established regulatory framework for these compounds due to their continual release or inherent persistence.

8.2.1 Soil Interactions

When sewage sludge and animal slurries are employed as fertilizers or polluted water is used for irrigation, APIs are released into the soil environment. Then, soil organisms might absorb these ingredients (earthworms or microorganisms) or it might be absorbed by plants and make its way into the food chain. Predicting the uptake of pharmaceuticals (such as carbamazepine, diclofenac, fluoxetine, and orlistat) by terrestrial systems requires knowledge of both soil characteristics and pharmaceutical physicochemical qualities (Carter et al., 2016). The interior chemistry of soil and earthworms can be altered by the drugs, which could have larger ramifications for risk assessment. Poultry manure fertilization results in the buildup of chlortetracycline concentration (Warman & Thomas, 1981). Moreover, drug metabolites excreted by medicated livestock are broken down into active medications by bacteria in liquid manure. As a result of the application of such manure to agricultural soils, livestock microflora become multidrug-resistant. Consequently, these numerous drug-resistant strains enter the food chain (Berger et al., 1986).

Antibiotic concentrations in aquaculture sediments are found to be too high to prevent bacterial growth (Kummerer, 2004). The compounds used in fish farming can enter sediments straight from water without going through any form of purification process since resistant bacteria may be present in sediments as a result of the use of antibiotics in fish farming. Studies have shown that antibiotics are present and persistent in sediments beneath fish farms (Kummerer, 2004). Fluoroquinolones, sulfonamides, and tetracyclines have a strong adsorbing capacity and thus are easily able to accumulate in sediments. The widespread use of antibiotics to treat bacterial infections in the fish farming industry (aquaculture, marine culture, etc.) has been linked to the emergence of antibiotic resistance (Serrano, 2005).

8.2.2 Environmental Transformation

Biodegradation, photodegradation, and abiotic transformation activities (such as hydrolysis) can partially convert and mineralize APIs, lowering their concentrations in the environment (Alexy et al., 2004). The intensity of solar radiation, water depth, the makeup of organic matter, eutrophic conditions, latitude, seasonality, and the presence of contaminants all affect how much photodegradation occurs. However, the compounds produced as the result of photodegradation are not necessarily nontoxic; according to a study, acridine is a by-product of carbamazepine's photodegradation in artificial estuarine water. This metabolite causes cancer, mutagenesis, and toxicity (Chiron et al., 2006). Furthermore, a chemical to be photodegraded should be present in available form; for example, the antibiotic tetracycline, which is commonly used in animal husbandry, cannot be photodegraded due to its adsorption onto sediment (Tolls, 2001). But with a (pseudo) first-order elimination rate and a brief half-life of 1 h, the analgesic diclofenac can be easily and quickly destroyed through direct photolysis (Buser et al., 1998). Similarly, photolysis can be used in surface waters to break down propranolol, Ibuprofen, Metronidazole, Acetaminophen, and several other APIs (Carlson et al., 2015).

8.2.3 Microbial Transformation

Biodegradation is carried out with the native microbial population in the environment. In wastewater treatment systems (Helbling et al., 2010) as well as in the environment, many active pharmaceutical components undergo microbial-mediated interactions that result in the creation of various compounds. The biodegradability of APIs with various chemical structures grouped in the same therapeutic class is anticipated to vary because biodegradation requires enzyme processes in accordance with chemical structures (Oenesois et al., 2009). It would be assumed that microorganisms would multiply more rapidly as a result of using API chemicals at specific quantities as a carbon or energy source, leading to the breakdown of APIs. However, as API concentrations rise, it may impede the process of breakdown and become harmful to microorganisms (Onesios et al., 2014). Depending on the physicochemical characteristics of the compound and the circumstances surrounding the wastewater treatment, APIs may change during the process. APIs may undergo complete conversion, partial transformation into intermediary metabolites, or in certain cases, no transformation at all (Xia et al., 2005). The toxicity is not always eliminated by wastewater treatment, and a large number of transformation products with unknown toxicity and persistence may still be present in the final effluent as well as in receiving water bodies (Hughes et al., 2013).

8.2.4 Leaching

Untreated wastewater is used for irrigation in developing nations where there are no wastewater treatment facilities and water shortage is a growing danger to agriculture (Jiménez & Asano, 2008). In industrialized nations, reuse of treated wastewater for agriculture is growing. APIs enter into the soil by the reuse of wastewater for irrigation of agricultural field (Jiménez & Asano, 2008) application of sewage sludge as soil amendment or manure containing veterinary drugs (Kinney et al., 2006a).

Active pharmaceutical ingredients (APIs) present in tile drainage have possibility to enter the receipt aquatic bodies after being spreading of liquid municipal effluent to soils (Lapen et al., 2008), on the other hand, groundwater contamination with different APIs can be detected by adsorption constants (Yu et al., 2009). It is crucial to fully comprehend the fate of these APIs since, in addition to potentially negatively affecting soil health, their persistence and adsorption characteristics may also endanger groundwater (Hammesfahr et al., 2008; Liu et al., 2009a). As active pharmaceutical ingredients are organic in nature and their fate in soil effects that are more likely to reach the aquifers beneath the region are acquired by the application of wastewater at high irrigation rates over a long period of time.

8.3 Active Pharmaceutical Ingredients: Toxicological Profile

The beneficial effects of pharmaceuticals for health in humans are widely acknowledged; however, less is known about the negative effects of drugs on terrestrial and aquatic ecosystems. Unintentional and uncontrolled discharge of pharmaceutical compounds from different sources is found to pose a severe threat to the environment. Active pharmaceutical compounds pose a challenge to many main consumers in food webs, including plants and soil-dwelling creatures (Chubin et al., 2021).

The management of pharmaceuticals in the environment is difficult for a number of reasons. Lower doses of pharmaceuticals result in a response since they operate within the biological system. However, nontarget creatures are suffering negative health effects from exposure to or alteration of original compounds. Pharmaceuticals are stable so that they can reach and interact with the target molecules. They are designed to degrade slowly for maximum benefit. Additionally, its continual usage causes releases into the environment that outpace rates of breakdown. Also, the use of veterinary medications in agriculture and aquaculture is a significant cause of direct or indirect water body contamination (diffuse pollution).

Wastewater treatment plants process enormous amounts of home and industrial sewage that include human waste in order to combat the potentially dangerous effects of medicines in sewerage systems. The result of this procedure, sewage sludge (biosolids), is regarded as a nutrient-rich organic residue that can be recycled and used as fertilizer (USEPA, 2007). However, throughout the wastewater treatment process, the APIs may not always entirely change into nontoxic molecules (Xia et al., 2005). Depending on their lipophilicity or other binding possibilities, the wastewater treatment process may either entirely change, mineralize the APIs (Richardson & Bowron, 1985), or convert them to a more lasting compound (e.g., ionic bindings). Despite the fact that the harmful consequences of taking a variety of pharmaceuticals have long been documented in both humans and animals, little is known about what happens to nontarget organisms when they are exposed to similar drug combinations.

8.3.1 Soil Biota

Due to the use of treated water and/or sewage sludge, active pharmaceutical ingredients (API) are present in terrestrial systems with elevated human activity levels. In soils, there exist several heavy metals and polycyclic aromatic hydrocarbons (PAHs), two dissimilar types of pollutants that are the subject of public concern and research. The accumulation of APIs in soil and then in living organisms or the food chain still needs to be investigated in detail as little information is available. Because medications may be bioactive and/or effective at low concentrations, their discovery in soils and biosolids has raised concerns. Persistent antibiotics and their metabolites accumulate over time in agro-ecosystems and may be hazardous to soil microorganisms (particularly bacteria) and crops (Baguer et al., 2000).

Microbial populations may be impacted by the ecological effects of antibiotics on the environment. Antibiotic degradation metabolites demonstrate toxicity to microorganisms (Ge et al., 2010). Acute toxicity and microbial resistance are caused by leftover medicines in soil (e.g., diclofenac), and result in endocrine disruption (e.g., 17β-estradiol) (Thebo et al., 2017; Thelusmond et al., 2016). Some APIs (tetracycline, antibiotics, tylosin, and fluoroquinolones) have high adsorption coefficients, so they get eliminated from the soil system posing no harm to the larger soil fauna (Hamscher et al., 2005). On the other hand, methamphetamine (an illegal pharmaceutical substance) stimulates the dehydrogenase activity in soil microorganisms, although nitrification activity remains unaffected (Pal et al., 2013). Tetracycline exposure, however, inhibits the growth of nitrifying bacteria over the long term (Katipoglu-Yazan et al., 2015). Additionally, the organization of the microbial community, as well as the phylogenetic structure, resistance, and ecological function of the micro-ecosystem are altered when antibiotics are added to the soil and water environments (Diao et al., 2004). There have also been reports of pharmaceutical contaminants accumulating in earthworms at application sites; triclosan has the highest bioaccumulation factor, at 27, among these pollutants (Kinney et al., 2008).

8.3.2 Aquaculture

Many APIs are not completely eliminated or are only partially removed during wastewater treatment, contaminating groundwater and thereby impacting aquatic life at various trophic levels. Antibiotics, especially neomycin, trimethoprim, sulfamethoxazole, and enrofloxacin, rank among the most harmful substances for aquatic organisms (Gworek et al., 2019). Both newly isolated rainbow trout hepatocytes and Daphnia spp. are fatally affected by amphetamine sulfate. Cocaine damages DNA (Cyto-genotoxic effects), induces cellular death, and results in the development of micronuclei in zebra mussels' cells (*Dreissena polymorpha*). Morphine has an immune-toxic effect (exhibited by lipid peroxidation, phagocytosis reduction, and esterase activity) on freshwater mussels (*Elliptio complanata*). Different algal species show different levels of sensitivity to the antibiotic. Most antibiotics (amoxicillin, benzylpenicillin, sarafloxacin, spiramycin, tetracycline, and tiamulin) are toxic to cyanobacteria. *Microcystis aeruginosa* is substantially more responsive to antibiotics than *Selenastrum capricornutum*. Antibiotics in the aquatic environment can also have deadly effects on aquatic species' early phases of growth and reproduction, which translates into fear for the entire population (Kümmerer, 2009a). Fish tissue has also shown to contain several APIs, including fluoxetine, sertraline, norfluoxetine, and demethylextraline (Nikolaou et al., 2007). Antibiotic by-products are possibly more bioactive, toxic, persistent, and mobile in the environment than their parent chemicals, which is a cause for concern. Because they may be converted back to the parent chemical, these offspring constitute a further source of pollutants (Carvalho & Santol, 2016).

8.3.3 Plants

Wastewater that is regularly generated worldwide as a result of human activity is treated using a variety of procedures in wastewater treatment plants. Massive amounts of sewage sludge that are unavoidably created as a result of the biological treatment of wastewater using the activated sludge technology need to be properly and safely disposed of. By applying biosolids to the ground, sewage sludge can be managed as waste while also receiving soils are amended. Although there are advantages, such as an increase in organic matter and nutrients, it has been demonstrated that applying biosolids to land in excess of the beneficially predicted levels may increase nutrient loss to surface waters (Singh & Agrawal, 2008). Receiving soils have been reported to contain an increased amount of a number of contaminants connected to wastewater (pathogens, heavy metals, toxic organic pollutants, and pharmaceutical ingredients (Zaleski et al., 2005; McLaren et al., 2007; Gottschall et al., 2010). Following their presence in soils, these pollutants may subsequently move on to agricultural plants, where they may harm the plants or enter the food chain. People and live animals could be exposed to more of those chemicals by eating infected crop plants (Sridhara Chary et al., 2008).

As mentioned, biosolids are notoriously high in APIs, and using them as fertilizers contaminates not only agricultural soils but also surface and groundwater, which causes accumulation of these chemicals in plants. Pharmaceuticals have been found in soils irrigated with treated water, often at concentrations between 0.02 and 15 g kg^{-1} (Kinney et al., 2006). According to recent research, reusing biosolids may allow pharmaceutical compounds to be transferred into plants (Karnjanapiboonwong et al., 2011). Chemicals with varying physicochemical characteristics make up the active pharmaceutical components. Since many APIs are ionizable, their behavior during plant uptake may be significantly impacted by soil solution dissociation. Chemicals can react differently in ionized state compared to their neutral form, making this impact challenging to anticipate. As a result, it is critical to comprehend these processes to make better predictions about how APIs will behave. The observed harmful effects on plants were typically mild when exposed to individual API chemicals (Migliore et al., 2003). However, APIs are always a blend of several substances when they enter agroecosystems. According to a study comparing the effects of individual APIs and mixtures, exposure to a single API was shown to be less harmful than it actually was and did not allow for the prediction of the risk of combinations at environmentally relevant concentrations (Fernandez et al., 2013). Furthermore, Christou et al. (2016) hypothesized that the uptake pattern of APIs in a combination was distinct from that of APIs exposed singly. A study demonstrated that cucumber roots continuously accumulated APIs to a higher level than shoots when exposed to mixed APIs. Shoots showed a somewhat different pattern of individual API relative accumulation from that of where the highest concentrations of carbamazepine and diazepam were observed, followed by meprobamate and trimethoprim (Wu et al., 2013). The contrasting development environments, plant cultivars, and sampling intervals between the experiments could be to blame for the disparity.

According to research by Liu et al. (2009b), tetracyclines and sulfonamides have been demonstrated to hinder seed germination, albeit the effects varied depending on the plant species and the various PPCPs taken into account in the study (Carvalho et al., 2014).

The bio-concentration factor (BCF), which was developed by Esring and Turan (2012), or bio-accumulation factor (BAF), which was developed by Sadiq (1992), is used to measure the accumulation of various compounds in organisms. The net uptake of a pollutant from the environment (soil or water) is estimated using the relationship between environmental pollutant concentrations and pollutant tissue concentrations as:

$$BCF = \frac{\text{Element concentration in plant}}{\text{Element concentration in soil}}$$

$$BAF = \frac{\text{Element concentration in plant}}{\text{Element concentration in Water}}$$

BCF values greater than 1 indicate that the accumulation in the organism is greater than that of the media (such as soil or water) from which the pollutant was extracted. These variables may be computed for the entire organism or standardized to the organism's lipid content (Eggen et al., 2011). The interaction of APIs with the substrate, their interaction with pathways, and their physicochemical characteristics like pKa and Kow all affect how well they are absorbed by plants. Concerns about APIs' phytotoxicity arise from their potential to infiltrate plants. At environmentally relevant doses, some medicines have been found to have detrimental effects on plants (Aristilde et al., 2010). Humans and other organisms that consume plant components may be at risk from accumulating APIs through the food chain. APIs may be metabolized after being taken up by the plant root, which results in their detoxification, deactivation, or accumulation (He et al., 2017). Reactive oxygen species (ROS) production is another pivot position as in response of plant cells to toxins, ultimately placing oxidative stress on essential plant macromolecules (Ahammed et al., 2012). Recent research revealed that longer-term visual phytotoxic reactions including seed germination reduction and root growth inhibition may be caused by oxidative damage triggered by ROS overproduction (Mittler, 2002). Increased ROS generation, a direct consequence of stress-induced cellular changes, places biomolecules under oxidative stress (Gill & Tuteja, 2010). Indicating the existence of oxidative stress, the cucumber plants' increased production of ROS also increased membrane deterioration and lipid peroxidation. However, alfalfa leaves did not significantly oxidize as a result of cucumber exposure to 10 mg L^{-1} of diclofenac, sulfamethoxazole, trimethoprim, or 17a-ethinylestradiol (Christou et al., 2016). A rainbow trout gonadal cell line was found to be more cytotoxic after exposure to mixed PPCPs than after exposure to individual API (Fernandez et al., 2013). These findings suggested that research employing PPCPs individually may have overestimated the true environmental effects of trace organic pollutants, which typically exist in combinations. Cucumber plants were exposed to the API mixture for the specified amount of time, and it was found that both enzymatic and nonenzymatic antioxidant systems were generally induced. The initial line of defense against

ROS, which can dismutate O_2 into the more stable H_2O_2, is superoxide dismutase (SOD) (Mittler, 2002). Greater ROS formation caused more serious oxidative damage to the root than to the shoots and higher SOD activity were seen in APIs treatments below 5 mg L^{-1}. The level of SOD activity in roots was dramatically suppressed above the 5 mg L^{-1} treatment rate (Chengliang et al., 2018). An et al. (2009a, 2009b) discovered that paracetamol exposure boosted SOD activity, whereas triclosan and galaxolide stress resulted in lower SOD activity in wheat seedlings. The fact that the biochemical mechanism was unable to cope with the oxidative stress may be one explanation for the lower SOD activity. It should be emphasized that there are three different types of SOD isoenzymes, each of which is found in a different cellular compartment. These include copper/zinc-containing SOD, manganese-containing SOD, and iron-containing SOD.

According to Noctor and Foyer (1998), glutathione is one of the most important soluble low-molecular-weight antioxidants and the principal nonprotein thiol in plant cells, helping to maintain cellular communication and redox equilibrium. Additionally, the conjugation of GSH with xenobiotics may be a typical mechanism for the metabolism of different synthetic compounds in plants (Neuefeind et al., 1996). It has been demonstrated that diclofenac, chlortetracycline, and 8:2 fluorotelomer alcohol conjugate with glutathione in plants (Zhang et al., 2016). It is well known that these processes call for the extensive usage of reduced GSH as an electron donor, which leads to the production of oxidized glutathione (GSSG). Changes in the cellular glutathione reserve, especially the ratio of reduced to oxidized glutathione that is associated with these changes, play a critical role in plant defense mechanisms (Mittler, 2002). However, the effects of exposure to APIs at trace levels on glutathione homeostasis have not yet been fully studied. While the glutathione concentration declined to average limits at the maximum APIs treatment level, it increased at low APIs doses. When the APIs concentration was low (5 mg L^{-1}), the GSSG content remained steady, but when concentration was raised to 50 mg L^{-1}, GSSG significantly rose (Chengliang et al., 2018). The oxidative stress and xenobiotic detoxification caused by APIs may be facilitated by the reported reduction in cellular GSH. The differing GSH responses in the roots and leaves, meantime, suggested that the roots may be the primary site for APIs' toxicity expression and API detoxification induction. Given that GSH conjugates predominate in the removal of pesticides, this is likely that GSH conjugate will also remove trace pollutants like APIs. Therefore, it is essential to carry out additional study to examine the processes and pathways of GST-mediated phytotransformation of APIs following plant uptake (Chengliang et al., 2018). Hence, oxidative stress responses may be employed as early plant responses to minor organic pollutants delivered into agro-ecosystems and may reflect the severity of API exposure and vulnerability of plant species to APIs. Additionally, plants may detoxify APIs using a variety of methods, such as boosted cellular antioxidant to fend off oxidative stress and excitability of xenobiotic-metabolizing enzymes. These systems support the maintenance of the physiologic, biochemical, and cellular functional integrity of plants, opening the door to the potential application of some of these metrics as diagnostics for phytotoxicity prognosis brought on by APIs and maybe other synthetic compounds.

Through root absorption (Dolliver et al., 2007), active pharmaceutical ingredients (APIs) move from soil and water into plant tissues where they accumulate (Wu et al., 2012, 2013). Three antibiotics, sulfamethoxazole, trimethoprim, and ofloxacin, as well as an antiepileptic drug, carbamazepine, were studied for their ability to accumulate in common wheat (Triticum aestivum L.). Ofloxacin accumulates in grain and straw at rates of 2.28 and 0.89 ng/g and 10.2 and 7.05 ng/g, respectively. Trimethoprim only builds up on grain and straw surface, whereas carbamazepine and sulfamethoxazole are found at 1.88 ± 2.11 and 0.64 ± 0.37 ng/g, respectively (Franklin et al., 2016). According to studies, some APIs (trimethoprim @ 1.1 0.2 mg/g dry weight) can be absorbed by spinach and lettuce from soil watered with treated effluent. Lower amounts of trimethoprim and sulfamethoxazole have both been found. However, corn, lettuce, potato, cabbage, and green onions only contain modest amounts of chlortetracycline and sulfamethazine (2–17 ng/g) (Kumar et al., 2005). According to studies, all of the investigated plant parts (carrot and barley) had ciprofloxacin and narasin bioaccumulation coefficients that are less than 1. The coefficient of bioaccumulation of metformin, however, is often larger in the roots (2–0) than in the leaves (0.1–1.5) when it is detected in sewage sludge (@ 0.5–1.6 mg/kg dry weight). Additionally, it has been found that turnip seeds have a bioaccumulation factor that is 40–60 times higher than that of wheat or barley seeds, which is above 1.

When APIs concentrations were high, morphological markers were affected, and cucumber roots were more significantly affected than cucumber shoots. It is possible that the larger API buildup in the root is what causes the hypersensitivity of root system to API poisoning (Chengliang et al., 2018). After exposure to APIs, similar findings in alfalfa, lettuce, and pepper were previously observed (Christou et al., 2016).

Following this, there is a significant immediate risk associated with human ingestion of plants grown on manure-fertilized soils and having absorbed veterinary antibiotics (Halling-Sørensen et al., 1998; Kümmerer, 2009a, 2009b). Therefore, the danger may be significant when consuming substances from multiple sources at once, at extremely low daily permissible doses, or when elusive effects are produced over an extended period of time (Kümmerer, 2009a, 2009b).

Antibiotics in the soil may impact the soil fauna (micro-worms, nematodes, and protozoa), which in turn affects the activities taking place in the soil, affecting the biological balance of plants. Plant wastes that decompose slowly release fewer nutrients into the soil over time due to a lack of denitrification processes (Fatta-Kassinos et al., 2011). The soil's qualities, such as its organic carbon content and ion exchange capacity, affect the mobility and bioavailability of APIs to plants (Carvalho et al., 2014).

8.4 Strategies for Remediation of Soil Contaminated with Active Pharmaceutical Ingredients: Prospects and Challenges

With the introduction of new medications to treat previously incurable ailments as well as newly invasive diseases throughout the past century, pharmaceutical use has been rising and will continue to do so in the future (IMS Health, 2012). Since most active pharmaceutical ingredients (APIs) are resistant compounds that are frequently present in tiny concentrations in complicated mixes, removing them from wastewater is a rigorous and challenging task. The enormous range of contaminants' characteristics with their various chemical structures and functions, is in reality, a significant complicating problem. Due to the variety of APIs, there must be a wide range of legal, disposal and treatment strategies that are not only efficient but also technologically and financially viable. Creating a layout of current domestic and international pharmaceutical activity is the most crucial step. Prioritize actions requiring appropriate expertise in monitoring, guiding, and training tools. Good Manufacturing Practice (GMP) should be used to integrate environmental quality standards. The healthcare industry should improve its use of green procurement. The administration of veterinary medicine should be replaced with nonchemical alternatives in animal husbandry and aquaculture. It is important to choose technologies that are adequate for managing pharmaceutical waste, such as incineration facilities. Additionally, facilities for cutting-edge sewage treatment and hygiene should be built. Drug take-back programs, for instance, should be created and promoted to ensure the proper administration of collecting and disposal schemes for the disposal of unused/expired prescriptions. Additionally, facilities (such incinerator facilities) that are adequate for managing pharmaceutical waste should be constructed (Weber et al., 2014).

Increasing public awareness of the harmful effects of pharmaceuticals entering the environment will help to improve the knowledge and information system about pharmaceuticals and will have an impact on prescription, usage, and disposal patterns (e.g., by discouraging people from flushing unneeded medications down the toilet) (Tong et al., 2011). A global network of scientists and risk managers should be established to ease information exchange, and a state-of-the-science report should be requested from the International Program on Chemical Safety (IPCS) to provide decision makers with the most recent knowledge. When environmentally friendly alternatives to a drug are available, information on those should be made readily available to inform prescription, purchase, and usage decisions (Weber et al., 2014).

Improving the coordination and synergies of ongoing actions at the global, regional, and national levels is the most crucial step in controlling API contamination (e.g., the Joint UN Program of Green Procurement in the Health Sector; the WHO program on quality and safety of medicines; relevant SAICM initiatives, such as EDC Strategy, as well as other existing regional and national initiatives). Consideration should also be given to industry consultation. Additionally, setting limitations or thresholds for pharmaceuticals with ecotoxicological relevance in surface waters can be beneficial for waste management. Furthermore, illegal traffic of pharmaceutical should be controlled so that actual production and usage of pharmaceutical may be monitored (da Justa Neves & Caldas, 2017). To address the most serious problem of API pollution, developing nations and nations through economic transition require capacity building and technical cooperation. As a result, monitoring programs, established procedures, and analytical tools that can detect pharmaceuticals in environmental matrices at appropriate quantities can aid in minimizing the effects of APIs on the environment.

The treatment of emerging contaminants has been projected to use a variety of approaches, such as membrane filtration, enhanced oxidation technologies, physicochemical therapies, and biological treatments (Acero et al., 2013; Secondes et al., 2014; Kim et al., 2018; Serna-Galvis et al., 2019). Numerous cutting-edge approaches to treating domestic water have been proposed, including biosorption onto unconventional materials (Li et al., 2019; Fenyvesi et al., 2020); filtration membranes combined with a biological step; algal-based technologies; and oxidation processes combined with an adsorption pretreatment (Rodriguez-Narvaez et al., 2017; Serna-Galvis et al., 2019). Recent research suggests that increased oxidation techniques, such as ozonation, electrochemistry, photocatalysis, sun-driven processes, or solar photo-assisted H2O2 treatment, may be used to prevent APIs (Polo-López et al., 2014; Serna-Galvis et al., 2016, 2019). Modern oxidation techniques have unquestionably been proved to drastically lower the load of microorganisms and organic pollutants in contaminated water (Moreira et al., 2016). This is important because wastewater that has been adequately treated can be utilized as a final product, such as in agriculture, and can take the place of renewable water sources. Industrial water can also be treated using these methods (Wang & Wang, 2016; Crini et al., 2019).

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Chapter 9
Physiological and Molecular Mechanism of Nanoparticles Induced Tolerance in Plants

Asad Azeem, Nasir Abbas, Sonia Azeem, Zafar Iqbal, and Sami Ul-Allah

Abstract Nanoparticles are emerging plant contaminants applied through soil or foliarly to deliver plant nutrients to the plants for growth development and stress tolerance. Nanoparticles capable of entering into plant cells and leaves can transport nutrients into different parts of the plant. Nanoparticles contain magic bullets such as nano-fertilizer and nano-pesticides. Naturally occurring nanoparticles are found in volcanic ash and ocean biological matter such as viruses and dust. The prime application of nanotechnology is to increase crop production with minimum losses and to activate plant defense mechanism against pests, insects and other environmental challenges. In this chapter we will discuss nanoparticles, their fate in plants, and their role in physiological and molecular stress tolerance in plants.

Keywords Nanoparticles · Crop nutrients · Abiotic stress · Nanotechnology in agriculture

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A. Azeem · N. Abbas
Department of Plant Breeding and Genetics, University of Agriculture, Faisalabad, Pakistan

S. Azeem
Department of Zoology, Government College University Faisalabad, Layyah Campus,
Layyah, Pakistan

Z. Iqbal · S. Ul-Allah (✉)
College of Agriculture, Bahauddin Zakariya University, Bahadur Sub-campus,
Layyah, Pakistan
e-mail: samipbg@bzu.edu.pk

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9.1 Introduction

Plants have to face a diversity of environmental intimidations from germination to maturity. They evolved several protective mechanisms at physiological, biochemical as well as at molecular level to address all kinds of fluctuations in terms of environmental threats (Laware & Raskar, 2014). Multiple losses of yield reduction of field crops have been observed owing to abiotic stresses in plants (Gilliham et al., 2017). Excess of salt, drought, elevated temperature, and alkalinity are major abiotic stresses that play a principal role in the reduction of plant performance and ultimately plant yield. In many cases, excessive salt concentration in soil and drought are the major yield-limiting factors of field crops (Jamil et al., 2011).

Excessive salt concentration in soil has become a leading issue in research and field crop production. A report on salinity showed that about 20% of the world's cultivated land is threatened by salinity. The soil is classified as a saline soil if the concentration of water-soluble salts in the soil is more than 4 dS m^{-1}. Owing to a sudden increase in the world population the agricultural lands have been replaced by human population, which is a leading factor of imbalance in food supply and food demand. To tackle this issue research into more food production and a limited supply of water on less cultivated area is being processed (Fischer et al., 2012). Ahmed et al. (2016) reported many negative impacts that have been observed on plants owing to excessive salt concentration, including disturbance in plant nutrient balance, reduction in soil osmotic capability, and salt toxicity in plants.

Like salinity, a limited supply of water is also a yield-limiting factor in crop plants. About 40% of the land is considered as dry land and is increasing owing to global warming (Thiry et al., 2016). Therefore, water is the prime abiotic factor in the reduction of crop productivity. Owing to a limited supply of water about a 36% yield reduction has been observed in crop plants. The reduction in crop yield falls within a wide range of diversity depending upon crops under stress and the severity of the stress as well as the stress period, To tackle all kinds of stresses, there should be plants with updated genetic make-up and their acclimation should also be borne in mind coupled with climate change (Wani et al., 2016). Many plants pathways are involved to address chronic abiotic stresses and to maintain normal growth and development even under stress condition.

The utilization of nanoparticles is an innovative and trending method of addressing plant stresses, especially abiotic stresses (de la Rosa et al., 2017). The application of nanomaterials in terms of nanoparticles is a basic component of nanotechnology. These nanoparticles are of nano-size ranging from 1 to 100 nm with a very high surface-to-volume ratio and reactivity. Nanomaterials vary in their size, type, composition as well as surface property and reactivity, which leads to change in physiological and biochemical properties from the remaining part of nanomaterials. Nanotechnology is one of the emerging fields for enhancing crop growth and development in advanced agricultural technologies. Nanoparticles play vital role in agrochemical production such as nano-pesticides, nano-fertilizer, and organic agriculture (Qian et al., 2020). Utilization of nanoparticles in agriculture is presented in Fig. 9.1.

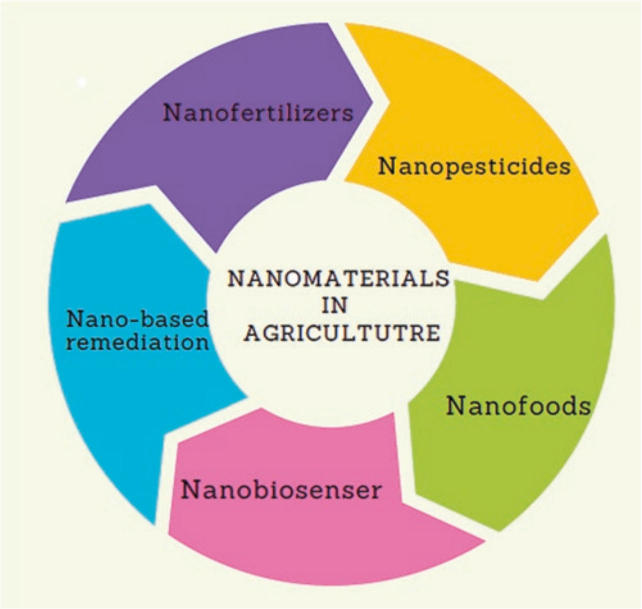

Fig. 9.1 Utilization of nanoparticles in the agricultural sector

The importance of nanoparticles cannot be denied in agri-products such as edible food production and nano-food items, biosensors, nano-biocomposites, and environment for agricultural crops, nano-biocomposites, biofuels, and in the postharvest management of crops. Nanoparticles also play a vital role in gene transmission for better nutritional value and for developing resistance against various types of plant stresses (Cunningham et al., 2018).

The application of nanoparticles in agriculture is the most essential tool in improving growth and resistance against plant stresses. In order to explore a variety of linkages between nanoparticles and plant resistance against stresses many studies have been conducted by scientists (Ali et al., 2021). Nanoparticles play an essential role in improving the germination of seed to seedling, including plant physio-biochemical activities such as photosynthesis, antioxidant enzyme activities, chlorophyll quality, carbohydrate content, and yield.

In addition, nanoparticles also a play significant role in highlighting the association and effects of stresses on numerous plant species through physio-biochemical and molecular mechanisms (Ali et al., 2021). These particles can also improve plant morphological features, biochemical features including enzymatic and non-enzymatic antioxidants, and gene expression associated with plant stresses (Silveira et al., 2019).

9.2 Fate and Behavior of Nanoparticles in Soils

Studies on nanoparticles showed their translocation, accumulation, and role in plants (Ali et al., 2021). According to studies, nanoparticle uptake, translocation, and accumulation vary from plant to plant because of changes in species owing to variation in nanoparticle size, type, reactivity, and interactions with the soil environment, root metabolites, and root-linked microorganisms (Fig. 9.2).

Soil acts as the most reliable nanoparticle receptor. The actions of nanoparticles, especially in risk evaluation, have become essential in soil. Soil as an innate medium comprises both primary as well as secondary nanoparticles (aggregates). The artificial introduction of nanoparticles into soil also has a considerable effect in increasing resistance against stresses in plants (Javed et al., 2019).

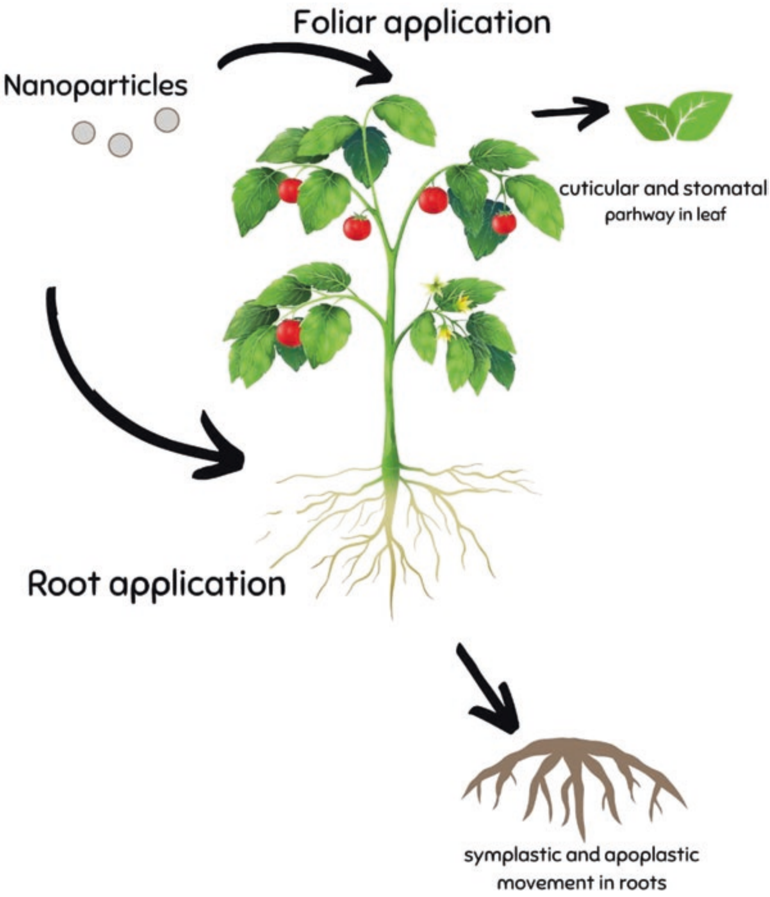

Fig. 9.2 Different ways of entry of nanoparticles into the plants

Nanoparticles with a small size are more mobile with high bioavailability compared with those of a large size. These nanoparticles have a tendency to accumulate, which is involved in the alteration of soil surface properties. The behavior of nanoparticles (NPs) in soil will control their actual movement and their bioavailability to soil organisms. The size scale of nanoparticles, composition, and charges are key to aggregation (Abbas et al., 2020). Sometimes, these aggregates are committed to stick with the soil medium and become relatively more constant. Therefore, the degree of aggregation directs the movement, chemical reactivity, and bioavailability in soil. Brownian diffusion is a nanoscale method that make sure the movement of nanoparticles in the soil environment. The rate of diffusion is directly proportional to the rate of collision, which leads to a diversity of association between sessile as well as mobile soil particles and nanoparticles (Lin et al., 2010).

The impression of nanoparticles depends upon the three factors: concentration of nanoparticles, soil property, and enzymatic activity. A variety of factors affect the fate and role of nanoparticles in the soil environment. As a result, the change in the properties of nanoparticles along with soil properties leads to a change in chemical and physical interactions, which ultimately influence the agglomeration as well as aggregation of nanoparticles (Tourinho et al., 2012).

Molleman and Hiemstra (2017) proposed the influence of soil pH on the role and behavior of nanoparticles. At low pH, nanoparticles quickly transform into cation or anion, but at high pH aggregation of nanoparticles was observed to be very high. Change in soil pH, either low or high, affects the function of changing charges on nanoparticles, resulting in alteration of the aggregation process (Dimkpa, 2018). For instance, titanium oxide nanoparticles possessing a negative charge are relatively more constant in nano-form under alkaline pH.

The behavior of nanoparticles can be altered as the salt concentration and strength fluctuate, resulting in alteration in the plant's performance. Soil organic matter mainly comprises organic acids and dextran, which constitute a number of compounds in soil as the concentration of organic matter varies in the soil the behavior of nanoparticles is also changed due to the change in surface charges on them (Dimkpa, 2018). Studies showed the negative correlation of cerium dioxide nanoparticles and organic matter in maize root; owing to the presence of organic matter the concentration of cerium dioxide nanoparticles was reduced.

An abundant supply of phosphorus has also changed the fate of nanoparticles in numerous cropping systems. Phosphate has the ability to bind with iron, copper, and zinc, therefore resulting in a reduction in their bioavailability. For instance, synthesis of insoluble zinc phosphate aggregates resulting from the dissolution of zinc ions from zinc oxide nanoparticles, which is precipitated with phosphate. Thus, the rate of zinc phosphate synthesis is directly proportional to the phosphate concentration and inversely proportional to zinc ion dissolution (Lv et al., 2012). The effect of different nanoparticles on plants has been presented in Table 9.1.

The fate and behavior of nanoparticles also depend upon soil biology. Studies depicted that absorption of silver nanoparticles in *Trifolium repens* plant roots was disrupted by mycorrhizae and varied in different species (Feng et al., 2013). Soil biological activity is also assessed by exoenzyme activities. Some significant soil

Table 9.1 Positive response of nanoparticles to various plant species under stress

Nanoparticles	Nanoparticle size (nm)	Nanoparticle concentration	Stress	Plant species	Response	References
Zinc oxide nanoparticles	30	10 mg L^{-1}	Sodium chloride 6000 mg L^{-1}	*Brassica napus*	Oxidative defense hyperregulation, and more osmolyte synthesis	Farouk and Al-Amri (2019)
Carbon nanoparticles	6–12	20 mg L^{-1}	125 mM sodium chloride	*Brassica napus*	Tolerant against sodium chloride stress	Zhao et al. (2019)
Carbon nanoparticles	80–100	50 mg L^{-1}	100 mM sodium chloride	*Sophora alopecuroides*	Increase plant metabolism, energy production and glycolysis	Wan et al. (2020)
Iron sulfate nanoparticles	90	2 g L^{-1}	100 mM sodium chloride	*Helianthus annuus*	Promotes antioxidant activities	Torabian et al. (2018)
Copper nanoparticles	50	250 mg L^{-1}	50 mM sodium chloride	*Lycopersicon esculentum*	Promotes antioxidant activity, and plays a role in sodium potassium ratio	Pérez-Labrada et al. (2019)
Silver nanoparticles	15–29	0, 2, 5 and 10 mM	25- and 100-mM sodium chloride	*Triticum aestivum*	Promotes sugar, POD activity and proline content	Mohamed et al. (2017)
Pristine MWCNT	Not specific	20 mg/L	Zero stress	*Zea mays*	Nutrient transport, biomass	Tiwari et al. (2014)
Iron sulfate nanoparticles	Variable	Variable	Water stress at critic stages	*Carthamus tinctorius*	Negatively affect drought	
Graphene	Variable	200 μg mL^{-1}	Drought	*Gossypium hirsutum*	Positive effect on fiber biomass	Pandey et al. (2019)
Titanium dioxide nanoparticles	Variable	0.01–0.03%	Drought	*Triticum aestivum*	Improvement in starch and gluten	Jaberzadeh et al. (2013)

factors are ionic strength, organic acids, and soil biochemical processes also have an impact on nanoparticle activities. The soil that has the abovementioned qualities is considered good for the stability, transport, and bioavailability of nanoparticles. Therefore, the measurement of these qualities is beneficial for the detection of the interaction of soil and nanoparticles in terms of bioavailability and toxicity.

Owing to the lack of a proper protocol and procedure for the evaluation of nanoparticles, the behavior and function under different conditions is not characterized. Detection of the aggregated state of nanoparticles and their interactions is done by microscopy based on atomic force or election transmission microscopy. Therefore, it is so easy to find out risks regarding the nanoparticles and their interaction with soil particles too.

9.3 Translocation, and Deposition of Nanoparticles in Plants

Plants are exposed to nanoparticles in two basic ways: one is foliar application, the other is basal or root application. In the case of foliar application, there is a barrier of cuticle, which is a natural shield for water loss prevention and overall plant leaf protection, but along with these functions there are two basic pathways for solute uptake: one is the lipophilic pathway, which permits only nonpolar solute, and the second is the hydrophilic pathway, which permits only polar solute. These pathways have some suitable sizes ranging from 0.6 to 4.8 nm (Lv et al., 2019). However, the nanoparticles measuring <4.8 nm would penetrate the leaf via the cuticle. But studies showed that the occurrence of nanoparticles measuring >5 nm were observed in the leaf with no scientific elaboration until now (Lv et al., 2019).

Many studies have been conducted to confirm nanoparticle entry pathways in different plants species by using transmission electron microscopy and confocal laser scanning microscopy. As nanoparticles enter into the leaf these are transported via the vascular system into the plant. The phloem vessels play an appropriate role for the translocation of nanoparticles to the root.

In taller plants, there are two basic mechanisms for root absorption and transfer of nanoparticles. In many cases, the absorption of nanoparticles occurs though underlying pores, then they pass through the cell wall and ultimately into the intercellular space. The second mechanism of absorption and transfer of nanoparticles is a cell-to-cell transfer pathway in which the first step is penetration into the cell and then merging into the cytoplasm of the cell after these nanoparticles have been transported via plasmodesmata to other cells. In this mechanism only xylem vessels play an appropriate role in the translocation of nanoparticles from root to root (Lv et al., 2019).

9.4 Effects of Nanoparticles

Nanoparticles enter the plant system through several ways, the most common of which are the roots and leaves. After entering the plant system, nanoparticles interact with the plants at cellular and subcellular levels, leading to the morpho-physiological variations (Khan & Bano, 2019). This interaction could be useful or harmful based on the kind of nanomaterials and plant species. The influences of nanoparticles on plant systems might be defined by their chemical composition, reaction rate, size, and particularly concentrations of nanoparticles already present in the plant and artificially applied concentrations through irrigation or foliar spray. The investigations have made it clear that nanoparticles have a positive effect on seed germination and plant growth using a variety of NP delivery methods, which include soil application, foliar spray, and seed treatment at doses below certain limits. It also has been reported that nanomaterials have an encouraging influence on plant growth and development. Most of these studies were conducted under a controlled environment such as plate growth media and hydrophobic or pot environments. We discuss here only the good impact of nanomaterials on plant growth, seed germination, and plant stress tolerance to thoroughly understand the overall influence of nanomaterials on plants. Plants can respond to and survive in adverse environmental conditions including drought, salinity, cold, and heat stress. It has been comprehensively studied that those plants react to such abiotic stresses at the cellular and molecular levels. Plants' primary response to abiotic stresses involves a brief activation of cytoplasmic Ca^{2+}, ROS, AA, and increased MAPK pathways (Zhang et al., 2020). In the cutting-edge stage of stress response, the expression of genes and proteins associated with stress involved in cellular damage prevention are overexpressed. Nanomaterial engineering developments suggest that nano-fertilizers might be able to enhance agricultural outputs in the current prevailing unfavorable environmental conditions. On around 23% of the world's farmed areas, salt stress limits crop productivity. SiO_2 nanoparticles improve seed germination and enhance plant fresh and dry biomass, chlorophyll content, and proline deposition under sodium chloride stress in tomato cultivars. Iron sulfate ($FeSO_4$) nanoparticles applied foliarly enhance salt stress tolerance in sunflower genotypes. The use of iron (II) sulfate ($FeSO_4$) nanoparticles not only improves leaf surface area and the gross absorption rate of CO_2, it is also reported that $FeSO_4$ nanoparticles increase chlorophyll content, the dry mass of the shoot, and the concentration of CO_2 in substomata, which have a particularly positive effect on photosystem II (Fv/Fm) activity. These nanoparticles also enhance the iron content coupled with decreasing the significant amount of sodium in leaves. Titanium dioxide (TiO_2) nanoparticles upregulate the antioxidant enzymes in onion plantlets, and the studies also reveal that the antioxidant enzyme activity is enhanced when titanium dioxide nano-fertilizers were used combined with silver nanoparticles. At a lower titanium dioxide dose, the seed sprouting and onion plantlet growth are improved, but on the other hand, a larger concentration has negative effect (suppressed). Furthermore, at a low TiO_2 dose (10–30 g/mL), hydrolysis enzyme (amylase), catalase, and peroxidase activity are increased, but superoxide dismutase activity is increased by increasing the concentration in wheat. The use of titanium nanoparticles similarly reversed the detrimental impact of drought in wheat, which ultimately improves the agronomic characters such as starch and gluten at higher concentrations (40 and 50 g/mL) of TiO_2 concentrations (Laware & Raskar, 2014) The results showed that the use of 0.02% of TiO_2 nanoparticles improves various agronomic parameters such as yield, plant height, number of spikes, spike mass, number of grains per plant, plant fresh mass, harvest index, and starch concentration. To some extent, silicon can be used to overcome the impact of drought on plants. Early studies indicate that a lower Na_2SiO_3 dose of about 1.0 mM could recover wheat yield loss due to drought. Although the underlying process is still unclear, Si enhances shoot growth, increases chlorophyll content, and increases water retention capacity in stressed plants. Furthermore, Si prevents membrane oxidation reaction with lipids in wheat. It is reported that Si nanoparticles can dramatically decrease UV-B-induced wheat stress. The foliar application of nano-Si at a dosage of 2.5 mM potentially promotes resistance against cadmium in rice cultivars by limiting the cadmium accumulation. Nano-Si fertilizers tend to be superior to conventional fertilizers in terms of heavy metal accumulation (Khan & Bano, 2019). Despite extensive study on the influence of nanomaterials on plant growth and resistance, the exact mechanism mostly remains unclear. The use of nanoparticles such as zinc oxide or nano-SiO_2 increases proline and amino acid accumulation, water, nutrient absorption capacity, and antioxidant enzyme activity such as catalases, glutathione reductase, nitrate reductase, superoxide dismutase, and peroxidase; all these advance the tolerance of plants to extreme levels. Additionally, nano-fertilizers have a strong effect on the expression of stress-associated genes. The use of silver nanomaterials in *A. thaliana* microarray study reveals that a few genes that play a vital role in stress tolerance are upregulated or downregulated. The upregulated genes are those that express in enhanced metals and oxidative stress. In contrast to upregulated genes, the downregulated genes such as systemic attained tolerance, signaling of ethylene, and auxin-associated genes involved in the development and size of organs are associated with response to infections and hormonal stimuli (Zhang et al., 2019). Certain nanomaterial-induced reactions in plants play a direct role in plant defense against a variety of environmental stressors. The reaction of plants to nanomaterials differs with the kind of nanoparticles, plant species, and the plants' stage of growth (Zulfiqar et al., 2019). Further study is required before employing specific nanomaterial techniques to recognize signaling pathways and expression of a gene regulated through the specific nanoparticles in diverse crop plants.

Farmers rely significantly on pesticides to curtail yield losses, which has a heavy impact on the sustainability of the environment and human health. Current study has proven that nanoparticles can progressively decrease the threat of pests and pathogens, thereby minimizing the prevalence of crop losses and environmental concerns.

Zinc oxide, copper oxide, and magnesium oxide nano-fertilizers could be used to control pest and soil-borne pathogens in many crop species (Guerrero et al., 2020). Titanium oxide nano-fertilizers have the potential to minimize the harmful impact of chilling stress by reducing plant membrane rupture and ion leakage. Photosynthesis is an essential plant mechanism that is very sensitive to cold stress. Chilling stress degrades the photosystem enzyme (RuBisCO) and lowers the chlorophyll level, CO_2 absorption, and the transpiration rate. Nanoparticles can influence the photosystem of plants and increase RuBisCO enzyme production (Ze et al., 2011). Titanium oxide nanomaterials upregulate the RuBisCO and chlorophyll-binding protein genes, leaf pigments, and antioxidant enzyme activities, as well as tolerance to chilling stress.

9.4.1 Plant Physiological Activities Under Stresses

Plants are frequently exposed to a number of abiotic stresses. Modified plants are those that are successful in modifying various response mechanisms in favor of their protection in order to withstand such stresses and return to normal vital metabolism. Notably, these environmental limitations substantially limit agricultural development and production. For example, an intensification in atmospheric CO_2 can cause variations in the photosynthesis process of plants, leading to variations in growth rate, which usually has a positive impact on overall biomass but an adverse effect on nutritional quality. Plant responses to stress-induced signals bring about major changes in their development and physiology. Many mechanisms that involve photosynthesis and gas exchange (Bryant et al., 2021), cell death, modifications in cell-wall composition, nutrient remobilization, gene transcriptional activity, transposable elements, lipid signaling, metabolites, proteins, and antioxidant characteristics are changed because of the stress. The physiological response mechanism to stresses is a multifaceted sequence of events that begins with stress and concludes with different levels of physiochemical and developmental responses. In response to the abiotic obstacles described above, plants transmit their energy and nutrients to reproductive and defense systems, leading to decreased growth rate and lower vegetative growth. Various mechanisms of nanoparticle-induced stress tolerance in plants are presented in Fig. 9.3.

Plants are subjected to a diverse range of environmental conditions. Water scarcity, for instance, can have a severe effect on plant growth, resulting in decreased outputs. Plants have advanced a variety of adaptive and acclimation strategies with the goal of maintaining their hydric condition (Kapoor et al., 2020). The duration and intensity of plant responses are determined by the duration and harshness of water scarcity, and the plant's genotype, vigor, and developmental stage. Irrespective of how transient the stress is, the plant's assortment of features and important functions reduces over time. A small water deficit induces considerable morphological and physiological changes, but a severe water deficit may result in plant mortality. Drought resistance is a plant's potential to sustain and grow in a severe condition induced by drought. It is a good characteristic coupled with adaptability at multiple levels that let plants acclimatize in a certain environment to growth. Plant drought resistance mechanisms are divided into four categories: avoidance, tolerance, escape, and recovery. Drought avoidance is distinguished by a plant's extreme

Fig. 9.3 Mechanism of nanoparticle-induced stress tolerance in plants

ability to retain water in the event of a water shortage. This is capable of deforming plants together with cellular metabolism (for example, deep root system growth, stomata closing, leaf curling, and wax deposition on the surface of the leaf) (Perlikowski & Kosmala, 2020). Drought tolerance is dependent on plants' ability to sustain physiological activities in the face of acute drought stress by changing gene expression and metabolic pathways to avoid or restore damage resulting from stress. Drought escape is the process of coordinating a plant's development phase, life cycle, or planting period to escape drought during crop growth. Drought recovery refers to a plant's ability to restore metabolism after a period of extreme drought that results in complete growth, turgidity loss, and leaf desiccation. Drought avoidance or escape indicates lengthy drought circumstances, whereas drought tolerance refers to a brief yet severe drought period (Perlikowski & Kosmala, 2020). Plants, as previously noted, have adapted to water stress through a range of strategies ranging from anatomical to genetic alterations.

Over the previous half-century, land salinization has been shown to be a major bottleneck in agriculture. Now, salinization affects more than one-third of the world's agricultural land, reducing agricultural productivity and yield significantly. Now soil salinity is more serious at the international level owing to deprived irrigation systems, unsuitable pesticide usage, and industrial pollution (Zhao et al., 2020). Meanwhile, fluctuations in salinity levels in general might induce alterations in physiological and metabolic activity, restricting plant growth and development. Plants are stressed by salt in two ways: osmotic and ionic. A high amount of salt in the soil decreases soil water capacity, resulting in less water absorption through roots. An overabundance of Na^+ and Cl^- ions in plant cells, triggers toxification and nutrition issues (Ibrahimova et al., 2021). In this respect, it is mandatory to improve management approaches in order to limit the damage triggered by such events, as well as to promote research and plant development programs aimed at enhancing resistance to this phenomenon. With climate change and population increase estimates, cropland salinization is a serious future threat. Climate change will provide a barrier to crops soon in terms of development, growth, reproduction, and yield. Plants have evolved a variety of approaches to protect themselves from the damage induced by temperature increases, including chemical reactions and variations in their physiology and biochemical properties (Singh et al., 2019). These changes have damaged agriculture productivity.

9.4.2 Enzymatic and Non-enzymatic Antioxidant Activities Under Stress

Under stress, enzymatic and non-enzymatic activities are critical for maintaining an equilibrium among the synthesis and scavenging of reactive oxygen species (ROS). ROS were formerly seen to be harmful byproducts of aerobic metabolism. ROS has been shown to have a central signaling role in plants, effecting processes such as development, growth, especially plant response to biotic and abiotic stimuli in recent decades. ROS family members include free radicals such as O·2, OH·, and non-radicals such as H_2O_2 and $1O_2$. The primary sites of ROS production in plants are the chloroplast, mitochondria, and peroxisomes. The endoplasmic reticulum, cell membrane, cell wall, and apoplast are all secondary sites. Although the ROS family acts as signaling molecules in a variety of significant physiological processes, they also cause oxidative damage under a range of natural environmental stresses such as salt stress, drought, cold, heavy metals, and UV irradiation, whereas the fine balance among ROS production and eradication, which is required for successful cell homeostasis, is disrupted. It is reported by Ali et al. (2020) that cellular damage presents as the decomposition of macromolecules such as pigments, proteins, lipids, carbohydrates, and DNA, resulting in plant apoptotic cell death

9.4.3 On Growth and Overall Performance of Plants Under Stresses

Global climate change appears to be causing a variety of abiotic and biotic environmental concerns, influencing plant growth and productivity in general. Plants, like other living creatures, can protect themselves while being sessile in origin by developing numerous stress-resistance strategies (Bryant et al., 2021). They can withstand high temperatures, drought, flooding, salt, heavy metals, atmospheric pollution, toxic compounds, and an inclusive array of biological species such as, bacteria, nematodes, viruses, fungi, insects, arachnids, and weeds. Abiotic stresses may modify plant–pest interactions by rendering plants more vulnerable to pathogenic organisms. As an outcome, plants exhibit stunted growth, necrosis, and decreased chlorophyll content, as well as low photosynthesis efficiency, ultimately resulting in lower food yields (Uddling et al., 2018).

9.5 Molecular Mechanism

Plants are affected by numerous environmental stresses throughout their life cycle and, as a result, modify genetic, biochemical, and physiological pathways to strengthen their defense against environmental stresses in various phases. Plants respond to this type of abiotic stress by altering gene expression at the molecular level. Several research projects illustrated that the persuasive effect of nanoparticles on plant growth and development varies with dose. In plants, the signaling system excites the defense machinery, which activates molecular mechanisms to respond diverse stress circumstances. Calcium ions serve as a secondary messenger in signal transduction under various stress circumstances (Ali et al., 2020). The improvement of calcium ion levels in the cytosol caused by stress signals is predicted by calcium ion-binding proteins, which cause changes in gene expression and plant acclimation to stress circumstances (Khan & Bano, 2018). Nitric oxide (NO) has been found to promote the concentration of calcium ions in cytosol under diverse biotic and abiotic stress situations (Khan & Bano, 2018); consequently, calcium ions result in the synthesis of nitric oxide. In rice, the application of silver nanomaterials at the roots demonstrated that nanomaterials were included in responsive protein regulation and signaling of calcium ions, transcription, protein degradation, oxidative stress response pathways, cell wall formation, and cell division. It has been observed that under Al_2O_3 nanoparticle stress, miRNA genes are induced and play a crucial role (Singh et al., 2017).

9.6 Conclusion and Prospects

In recent decades, nanotechnology studies have proposed that the synthesis of nanoparticles could be a powerful strategy for reducing prevailing difficulties caused by conventional fertilizers in outdated agricultural systems. Nutritional material for plants consisting nanomaterial (i.e., Fe, Cu, Se, and Co), demonstrated a considerable scientific evidence of their effectiveness in enhancing crop micronutrients, as evidenced by upgraded growth parameters and significant physiological improvements (i.e., chlorophyll and carotenoids, photosynthetic activity, metabolic pathways, and transpiration rate). The precise amount and activity of nanoparticles in plants is a serious obstacle; therefore, reducing the seepage of chemicals obtained from bulk materials (i.e., mineral fertilizers) has become an appreciated characteristic that promotes the future usage of nanoparticles. In addition to nutritional stress, nanoparticles also play a role in the induction of tolerance to environmental stresses. Comprehensive studies under controlled environments, on the other hand, are critically necessary to assess the role of nanoparticles in the plant, as well as the fundamental nanoparticle dosage, acquaintance period, translocation and accumulation, and mode of action on plants are all critical considerations when developing an application strategy.

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Chapter 10
Arsenic and Cadmium Toxicity in Plants: Mitigation and Remediation Strategies

Gouranga Upadhyaya, Subhankar Mondal, and Aryadeep Roychoudhury

Abstract During growth and development, plants uptake a wide range of mineral nutrients from soil, including some non essential toxic elements such as arsenic (As), cadmium (Cd), etc. These contaminants inhibit uptake and translocation of essential solutes and nutrients as well as water, induce oxidative damage and disrupt plant metabolism. Such toxicity directly affects health of human individuals relying on such plants as a primary food source. As and Cd generally follow relatively similar pathway for entry in plant systems, as well as in terms of toxicological effects and sequestration machineries. They utilize certain common routes and membrane transporters for cellular uptake and translocation; however, some machinery also differs in this regard. Biomagnifications of As and Cd in the food chain is an important area of research to implement mitigation strategies. As a defense and detoxification mechanism, plants have evolved adaptive features such as efflux mechanisms, complexation with thiol-rich compounds and vacuolar sequestration. Plasma-membrane efflux transporters, phytochelatin-complexation and subsequent vacuolar sequestration have been reported to render As and Cd tolerance in important crop plants like rice. Although Cd levels are generally not high enough to cause phytotoxicity, Cd accumulation in edible plant parts is unsafe for human consumption. On the other hand, As-contamination in soil can cause massive phytotoxicity. Besides the traditional methods like soil amendments, use of soil microorganisms and plant growth regulators, modern-day-approaches like genetic engineering and/or priming techniques, especially with nanoparticles can significantly augment plant tolerance against metal stress and the associated biogeochemical calamities.

G. Upadhyaya
Department of Biological Sciences, Indian Institute of Science Education and Research Kolkata, Mohanpur, West Bengal, India

S. Mondal
ICAR-National Rice Research Institute, Cuttack, Odisha, India

A. Roychoudhury (✉)
Discipline of Life Sciences, School of Sciences, Indira Gandhi National Open University, Maidan Garhi, New Delhi, India

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This book chapter provides a comprehensive overview on agro-ecotoxicological aspects of the two emerging contaminants, namely, As and Cd along with adaptive mechanisms of plants and scientific approaches for metal stress alleviation.

Keywords Heavy metal stress · Cadmium · Arsenic · Bioremediation · Phytochelatin · Metallothionein · Nanotechnology · Genetic engineering

10.1 Introduction

Despite the fact that heavy metals like cadmium (Cd) and metalloids like arsenic (As) are biologically nonessential for the plants, they are taken up by the plants using the same transporters as for the uptake of essential nutrients (Verbruggen et al., 2009). As a result, As and Cd bioaccumulation and biomagnification through the soil, food chain and drinking water pose severe threats to human health and the environment. Along with natural geogenic processes like volcanic eruptions, forest fires, etc., anthropogenic activities by humans like metal/wood/electronic/plastic/rubber industries, fossil fuel combustion, use of contaminated fertilizers, herbicides, insecticides and irrigation with contaminated groundwater, sewage sludge, etc., altogether lead to the accumulation of high concentration of As and Cd in soil (Moulick et al., 2019). Hence, both Cd and As rank topmost on the US Agency for Toxic Substances and Disease Registry (ATSDR) Priority List of Hazardous Substances.

Once these toxic species gain entry into the cell, they can mimic the functions of essential metals and hence interfere with essential cellular processes. An element is usually classified as "essential" depending on its chemical nature and availability in the environment. As and Cd tend to fall in the "nonessential" category owing to their extremely low concentration in the Earth's crust (Roychoudhury et al., 2012a). In comparison to the abundance of phosphorus (P) and zinc (Zn), As and Cd lost their importance in the evolutionary race and were overpowered by P and Zn, respectively. Since As and Cd share neighboring positions with P and Zn in the respective columns of the periodic table, they tend to replace P and Zn, respectively, in various cellular metabolic processes, which turns the cellular atmosphere toxic (Tawfik & Viola, 2011; Tang et al., 2014). Therefore, plants have developed defense mechanisms to sense, transport and detoxify non essential metals and metalloids.

Many plant species have been found to show high tolerance level to the toxicity of As and Cd, and some plant species have shown exceptional ability to accumulate As (above 0.1%) and Cd (above 0.01%) in their shoot dry weight without any lethal indications. These plant species have been termed as hyperaccumulators (Manara et al., 2020). Only the Pteridaceae family of ferns to date have been identified to be As-hyperaccumulators, while many species of *Thlaspi*, like *T. caerulescens*, *T. praecox*, etc. along with others like *Arabidopsis halleri* (Brassicaceae), and *Sedum alfredii* (Crassulaceae) have shown Cd hyperaccumulation (Souri et al., 2017; Reeves et al., 2018; Tian et al., 2017). On the contrary, plants with heavy metal (As and Cd) sensitivity often show detectable symptoms like arrested growth, chlorosis, browning of roots, and may even result in death. Consequently, underlying metabolic, cellular and genetic mechanisms of the plants are often hampered beyond repair. On the other hand, plants with the ability to tolerate As and Cd toxicity have developed defense mechanisms for reduced uptake, vacuolar compartmentalization and sequestration through phytochelatins (PC)/metallothioneins (MT), along with activation of antioxidant defense (Verbruggen et al., 2009). Although there are similarities between As and Cd metabolism in plants, there exists significant differences as well, such as in transporters involved in these metal uptake and translocation, and their chelation techniques. Therefore, in order to remediate As and Cd contamination, several techniques have been implemented, starting with traditional soil constituent changes, priming the plants by different approaches and genetic engineering-based methodologies to fortify plant resistance to As and Cd toxicity (Roychoudhury et al., 2012b). Among the traditional option of attempting to remediate the heavy metal contamination from soil, physical, chemical, and biological methods can be integrated. Such most frequently used methods include thermal treatment, adsorption, chlorination, chemical extraction, ion-exchange, membrane separation, electrokinetics, bioleaching, phytoremediation (plants/fungal/algal-based), etc. (Selvi et al., 2019). Combination or integration of more than one treatment processes have been reported by multiple researchers worldwide to be more factually effective, and hence, these integrated processes have been gaining popularity for heavy metal removal from environmental matrices. Another approach to prime the plants towards acclimation to metal stress using wide variety of techniques has been reported to be quite successful, which underscores the fact that plant priming is among the most promising research fields of the near future (Wiszniewska, 2021). Priming with nanoparticles (NPs) is one of the most recently developed strategies with several advantages. For example, foliar application of different NPs lessened Cd intake by wheat plants and also improved growth, yield and nutrient uptake (Hussain et al., 2021). Moreover, use of genetic engineering to build plant resistance to As and Cd stress utilizes genes involved in transportation and detoxification techniques (Macnair et al., 1999; Bert et al., 2003). Development of transgenic plants, although an expensive technology, can prove most fruitful in the development of metal-free agriculture in the near future. Therefore, this chapter intends to highlight the similarities and disparities between As and Cd uptake, translocation and detoxification mechanisms in plants along with insight into the possible remediation strategies, like soil amendments, priming techniques and genetic engineering based-plant fortification.

10.2 Uptake and Translocation Mechanism of Arsenic and Cadmium in Plants—Similarities and Differences

Arsenic and cadmium present in soil is taken up by plant roots with the help of different transporters used for essential nutrient uptake. To date, several transporters involved in uptake and transport of heavy metals have been identified in multiple plant species. Depending on the redox status of the soil, As is usually found to exist in three main forms in soil, arsenate [As (V)], arsenite [As (III)] and methylated forms of arsenic [monomethylarsinic acid (MMA) and dimethylarsinic acid (DMA)], whereas Cd always exists in the divalent form (Awasthi et al., 2017; Kubier et al., 2019). Both these metal forms have been found to coexist in the soil, with many other metal constituents as well. Although particular bioavailability of these metal species is subjected to the soil structure, pH, organic matter and chemical speciation, Cd tends to be more bioavailable than As. Thus, the transporters involved in As and Cd uptake are different, as described under separate heads and represented in Fig. 10.1.

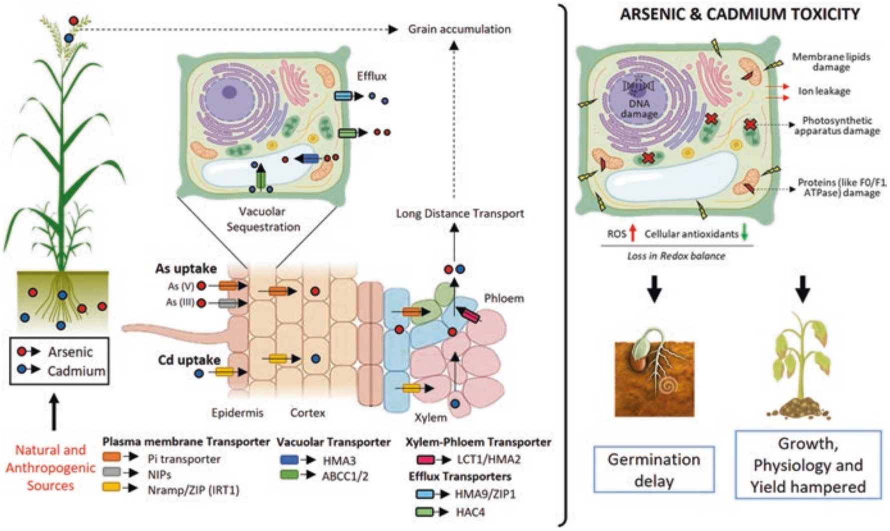

Fig. 10.1 Arsenic and cadmium uptake and transport, and impact on plant metabolism *Left panel*—Originating from both natural and anthropogenic sources, As and Cd accumulate in soil and are taken up by the plant roots. At the cellular level, As- and Cd both are taken up by the plasma membrane transporters in the cells of the plant roots, like Pi transporters helping in As (V) uptake; nodulin-like intrinsic protein (NIP) channels take up As (III) and NRAMP along with other ZIP transporters aid in Cd uptake. After traversing through the cortical zone, either by symplastic or apoplastic route, they are loaded in the xylem and phloem tissues with the help of certain transporters like LCT1/HMA2, which helps to transport them to long-distance organs, including grains (in case of rice or other cereals). As a method of detoxification, the zoomed-in image of plant cell above shows the transporters involved in vacuolar sequestration like HMA3 and/or ABCC1/2 along with the plasma membrane transporters helping in efflux of these heavy metals (HMA9 and/or ZIP1 for Cd and HAC4 for As). *Right panel*—It shows the toxicological impacts of As and Cd on plants at the cellular level that build up to affect plant growth and development, which ultimately affects the agricultural yield. At the cellular level, As- and Cd-stress can damage DNA, and proteins (F0/F1 ATPase, photosynthetic apparatus, etc.), inducing oxidative stress, lipid peroxidation, etc. Such severe cellular damages hamper the physiology and metabolism of plants and also delay the germination of seeds. The legend below depicts each of the transporters represented with separate coloured boxes

10.2.1 Arsenic Uptake and Translocation

All the different forms of As can coexist in soil, and can be also transformed from one form to another with the help of soil microorganisms, like As(V) is converted to As(III) and further to MMA and DMA. However, the different species of As are taken up by different transporters. First, As (V) is taken up into the plant root cells through the high-affinity Pi transport system owing to a high degree of structural analogy between As (V) and Pi. Many such high-affinity Pi transporters (Pht1; TC 2.A.1.9) have been characterized for their role in As(V) transport in plants. Examples include AtPHT1;1, AtPHT1;4, AtPHT1;5, AtPHT1;7, AtPHT1;8, AtPHT1;9, OsPHT1;1, OsPHT1;8, etc. (Li et al., 2016). In *Arabidopsis thaliana*, AtPht1;1 and AtPht1;4 have been reported for As (V) uptake under both low and high concentration of phosphorus in the soil (Shin et al., 2004). Another member, AtPht1;5 have been observed to promote Pi and As (V) translocation to sink organs, and hence, its knockdown mitigates As (V) toxicity in plants (Nagarajan et al., 2011). AtPht1;7 specifically overexpressed in the reproductive tissues of *Arabidopsis* enhances As (V) accumulation (LeBlanc et al., 2013). AtPht1;9 expressed in roots predominantly acts similarly to aid in As (V) uptake (Remy et al., 2012). Recently, multiple WRKY transcription factors like WRKY6 and WRKY45 have been also noted to act in As (V) influx (Castrillo et al., 2013; Wang et al., 2014). Studies in *Oryza sativa root and shoot tissues have identified OsPht1;8 with high affinity for both Pi and As(V), without the need of Pi supply. Overexpression of OsPht1;8 evidently enhanced As(V) uptake and translocation (Jia et al.,* 2011; Wu et al., 2011). Another constitutively expressed high-affinity Pht1 transporter involved in As (V) uptake was reported as OsPht1;1 (Sun et al., 2012). Besides the As hyperaccumulators, the As(V) hypertolerant plants contain suppressed high-affinity Pi uptake system leading to decrease in As(V) uptake. However, tolerance to As (V) does not ensure tolerance to As (III) as demonstrated in *Holcus lanatus*, because of the variance in uptake mechanisms of both the As species forms (Macnair & Cumbes, 1987). However, in an anaerobic environment, like rice growing in submerged condition, As (III) is the dominating species and as a result, the Pi transport pathway is unable to help in As-uptake (Xu et al., 2008). As (III) species is majorly taken up via the channels of the members of nodulin 26-like intrinsic protein (NIP) subfamily of aquaporin proteins (Li et al., 2014, 2016).The silicon transporter OsNIP2;1 (Lsi1) acts in As (III) influx into rice roots at the distal end of exodermis and endodermis, owing to molecular structure analogy of As (III) with silicic acid and this is why rice plants with high demand for silicon take up huge quantity of As from soil (Ma et al., 2006, 2008). Other transporters identified and characterized from Arabidopsis and rice include AtNIP1;1, AtNIP1;2, AtNIP3;1 (in roots) AtNIP5;1, AtNIP6;1, AtNIP7;1 (in developing anthers and pollen tissues), OsNIP1;1, OsNIP2;1, OsNIP3;1, OsNIP3;2 (in anthers and suspension cells), OsLsi6, etc. (Bienert et al., 2008; Isayenkov & Maathuis, 2008; Kamiya et al., 2009; Xu et al., 2015). However, in comparison to OsNIP2;1, OsNIP2;2 (OsLsi6), OsNIP1;1 and OsNIP3;1 are weakly expressed, implying the fact that probably they have a less significant role in As (III) uptake by rice roots, if any (Ma et al., 2008). Furthermore, OsNIP3;3 and HvNIP1;2 have been reported to show As(III) transport activity in yeast cells (Katsuhara et al., 2014). Yeast mutants showing absence of As(III) efflux transporter ACR3 (arsenical compound resistance 3), depict massive increase in the expression of OsNIP3;3 or HvNIP1;2 with increased sensitivity to As(III), establishing the fact that OsNIP3;3 and HvNIP1;2 act as As (III) transporters (Ali et al., 2012). In addition to NIPs, members of the rice plasma membrane intrinsic protein (PIP) family of aquaporins, including OsPIP2;4, OsPIP2;6 and OsPIP2;7, are also permeable to As(III) (Mosa et al., 2012; Samanta & Roychoudhury, 2021). Overexpression of these genes in Arabidopsis enhanced As(III) tolerance in plant roots without any long-term accumulation. Additionally, plants are also exposed to trace quantities of methylated As species, mainly MMA and DMA, due to the usage of pesticides or herbicides containing As or due to the As biomethylation by microorganisms (Ye et al., 2010). For example, rice grains have been observed to accrue twofold higher amount of DMA than inorganic As (Zheng et al., 2013). However, few studies to date have focused on the mechanisms of uptake and transport of methylated As. OsLsi1 (the aquaporin NIP2;1), known for As (III) uptake can also aid in the uptake of MMA and DMA by rice roots as shown by Li et al. (2009). The study showed that rice mutants without OsLsi1 show approximately 80% and 50% reduction in the uptake of MMA and DMA, respectively (Rahman et al., 2011).

10.2.2 Cadmium Uptake and Translocation

Cadmium is commonly taken up from the Earth's crust predominantly as divalent species (Cd^{2+}) by plant roots, while the chelated forms usually tend to be inaccessible for adsorption. Multiple factors like soil pH, humic acid and soil state can influence the effective Cd^{2+} concentration in soil. Moreover, the uptake of cadmium from soil into the root cells and its variable accumulation depends on several factors like concentration of Cd ions in the soil, Cd mobilization in the rhizosphere by root exudates, etc. Once the Cd ions get absorbed by the root hairs in the epidermal cells, they traverse successfully through the root cortex and the endodermis into the parenchymal cells, from where they are transported for long distances via the xylem tissues. To date, Zn transporters have been solely reported to play the most dominating role in the transport of Cd in presence of Cd concentration, being at least 100-fold greater than Zn, which in natural environmental conditions generally exists in the opposite ratio.

At the soil–root interface, the uptake of Cd^{2+} into the root can occur via three pathways: (i) by means of rapid exchange of H^+ (owing to plant respiration) with Cd^{2+} at the plasma membrane of root epidermal cells; this is an absolutely energy-efficient process occurring via apoplast pathway (Yamaguchi et al., 2011), (ii) through Fe^{2+}, Zn^{2+} and Ca^{2+} ion channels into the root epidermis through symplast pathway, and (iii) through yellow-stripe 1-like (YSL) proteins as chelates formed with mugineic acids (MAs) secreted by plant root exudates (Curie et al., 2009).

After travelling from the root exterior to the root cortex, including the apoplasmic barriers like casparian strip in the endodermis, Cd^{2+} either enters the symplastic route to get transported to the xylem elements, i.e., an apoplasmic space or directly gets delivered through the apoplasmic route into the stelar apoplasmic space (Redjala et al., 2009). Regardless of the symplastic/apoplastic pathway, loading into the root xylem is very critical for long-distance Cd transport, such as to the grains (Yamaguchi et al., 2011; Verbruggen et al., 2009). High Cd has been observed in the walls of xylem vessels in barley plants (*Hordeum vulgare* L.) and most of them exist in their chelated forms (Akhter et al., 2014). In addition to the xylem channels, phloem tubes also aid in long-distance source-to-sink transport of the chelated forms of Cd like with phytochelatins and glutathione (Mendoza-Cózatl et al., 2008).

Regarding the Cd transportation, Natural resistance-associated macrophage protein 5 (Nramp5) has been identified as a major Cd uptake transporter in the plasma membrane of root cells in rice (*Oryza sativa*) (Sasaki et al., 2012). Consequently, mutant *Osnramp5 from rice showed greatly reduced Cd uptake by roots with subsequent lower Cd accumulation in the straw and grain (Ishikawa et al.,* 2012). Although Zn^{2+} has been reported to inhibit Cd uptake in other plant species, it shows minute possibility to inhibit Cd^{2+} transfer in rice (Green et al., 2017). A very minute fraction might be retained in the apoplasmic space due to cell wall binding, while the majority is relocated via plasma membrane transporters for symplastic uptake. Cd has been shown to be taken up by MTP1, ZIP transporters (zinc-/iron-regulated transporter-like proteins) like IRT1 and nonselective cation channels in the plasma membrane (Yuan et al., 2012; Pence et al., 2000; Vert et al., 2003; Walker & Connolly, 2008). In consequence, cells can also export Cd in free forms and complex species from the cytosol through the plasma membrane via other transporters (Baekgaard et al., 2010; Migocka et al., 2011). Xylem-to-phloem transfer has been recently identified to be carried out by LCT1 (low affinity cation transporter) in rice plants, and its knockdown led to reduced Cd concentration in phloem sap of plants (Uraguchi et al., 2011). Therefore, the sequence follows uptake of Cd ions into the root cell symplast, retention in the roots through vacuolar storage, loading into the xylem, transfer and remobilization into the phloem leading to Cd accumulation. Once inside a cell, Cd can be sequestered within vacuoles through other transporters, as discussed in Sect. 10.3.

10.3 Agro-ecotoxicological Effects of Arsenic and Cadmium

Heavy metals largely accumulate in soil due to a number of factors like rock formation, industrialization, urbanization and synthetic use of arbitrary chemicals. Plants, being sessile, generally rely on their roots for food source and hence these metals, as contaminants, are taken up by plant roots as primary food knowingly or unknowingly (Shahid et al., 2015). The only exceptional case is aquatic plants, where their whole body gets exposed to the contaminants (Haider et al., 2021). As a result, As and Cd toxicity can affect almost all plant tissues throughout the life cycle starting from seed germination to senescence as shown in Fig. 10.1. This effect is more dominant in the root tissue for direct participation in their uptake and seed tissues owing to the final deposition (Ronzan et al., 2018). Taken together, these heavy metals decrease plant growth and development and result in massive decline in crop yield and quality. Such adverse effects in crop plants or other edible plants are a matter of concern, since the edible plant parts consumed by human population are heavily contaminated. On the other hand, no living organism can survive without the participation of metal ions. Some heavy metals (Fe, Cu and Zn) are critically essential as a source of micro and macronutrients (Zn, Fe, Cu, Mo, Mn, Co and Ni) (Singh et al., 2015). Their availability in medium or trace amount helps in several biochemical processes. They basically work in two modes: participation in redox reaction and aiding in the function of different enzymes (oxidases, carbonic anhydrase, superoxide dismutase, alcohol dehydrogenase, RNA polymerase, etc.) (Sharma et al., 2014; Singh et al., 2016). Interestingly, a large part of metal ions participate in photosystem integrity and functionality.

However, saturation of these essential metals (Cd and As) beyond the permissible limit of 100 mg/kg in soil results in excess uptake and toxic effect on plants, which reflects in visible injuries (Sun et al., 2021). Symptoms include chlorosis of leaves, wilting of whole plant, browning of root tips and finally death of the plants. The impact of As and Cd toxicity in plants can be distributed in different levels such as cellular, physiological, biochemical and morphological features, as described. Details of a few plants and their corresponding effects are listed in Table 10.1.

10.3.1 Seed Germination

Abscisic acid, gibberellin and auxin play the main roles during seed germination. It has been found that Cd can target their biosynthesis pathway and inhibit germination, growth and production. For cowpea (*Vigna unguiculata*) seeds, Vijayaragavan et al. (2011) showed that Cd can suppress water absorption, thereby minimizing the water supply to embryo seed. Another possible mechanism of delayed seed germination is inhibition of starch and soluble sugar mobilization from endosperm to embryogenic axis driven by these heavy metals (Kuriakose & Prasad, 2008). Calmodulin and calcium (Ca) play significant roles during seed germination, which can be hampered by Cd stress, as Cd competes with the Ca ions for the Ca-calmodulin binding sites (Huybrechts et al., 2019). During Cd stress, malondialdehyde (MDA) content was found to increase dramatically owing to Cd-responsive lipid peroxidation and delayed metabolic activation (Rahoui et al., 2010). In addition, As and Cd have been reported as oxidative stress inducers. They disrupt the balance between cellular hormones and ROS level, resulting in detrimental effect on seed viability (Jalmi et al., 2018). Interestingly, for both As and Cd, it has been found that a minimal concentration can stimulate seed germination rate (Hasanuzzaman et al., 2015; Haider et al., 2021). For example, lower concentration of As (0, 0.5, 1, 5, 15 and 20 mg kg^{-1}) on *T. aestivum* can increase the germination energy and germination

Table 10.1 Impact of cadmium (Cd) and arsenic (As) toxicity in different plants and role of different antioxidant enzymes combating oxidative stress

Metal	Stress type	Organism	Time	Effect	Reference
Arsenic (As)	100–500 µM	Rice	1–20 day	Higher H_2O_2 production and significantly reduced growth	Hsu and Kao (2007), Shah et al. (2001)
	50 µM	Pea	14 day	Oxidative stress generation; nitric oxide signalling stopped and degradation of cellular protein	Romero-Puertas et al. (2002), Rodríguez-Serrano et al. (2009)
	3 to 5 mM	Tobacco	30 min to 8 h	Induce oxidative damage, mitochondrial membrane disruption and higher NADPH oxidase production	Garnier et al. (2006), Olmos et al. (2003)
	50 to 250 µM	Wheat	24 to 48 h	Lipid peroxidation, electrolyte leakage and massive decline in root growth	Singh et al. (2008), Groppa et al. (2008)
	5 to 10 mg/L	Bean	4 day	DNA damage and lipid peroxidation	Lin et al. (2007)
	250 µM	Mustard	14 day	Cell death due to higher lipid peroxidation and inhibited growth	Guan et al. (2009)
	1 mM	Sweet potato	50 day	Reduced metabolic activity and plant growth	Kim et al. (2010)
	40 µM	Pea	7 day	Up regulation of APOX, CAT, GPOX, SOD helps to mitigate damages	Dixit et al. (2001)
	25 µM	Barley	10 day	Helps in chlorophyll accumulation by enhancement of APOX, CAT and GPOX	Metwally et al. (2003)
	220 ppm	Wheat	7 day	CAT up-regulation helps to mitigate stress. Enhanced seed germination and higher photosynthetic efficiency	Baruah et al. (2019)
	200 µM	Soybean	2 day	Improve nitrogenase activity and plant growth via enhancement in POX, CAT and SOD activity	Balestrasse et al. (2001)
	100–500 µM	Rice	20 day	Reduced MDA content and improved seedling growth by CAT, GPOX and SOD activity	Shah et al. (2001)
	2–5 mM	Sugarcane	4 day	CAT, GR and SOD helps to maintain photosynthetic efficiency	Fornazier et al. (2002)
	10 µM	Tomato	7 day	Reduced overall oxidative damage by APX, CAT, GR and SOD activity	Cherif et al. (2011)

(continued)

Table 10.1 (continued)

Cadmium (Cd)	50 mM As (III) and 500 mM As (V)	Rice	10 day	Reduced root formation, shoot length and fresh weight of seedlings. SOD, APX, POD and GR activity increased as line of defense	Shri et al. (2009)
	20 mg kg^{-1}	*Triticum*	7 day	Decreased germination index, vitality index and Chl content. Increased activities of SOD and APX but decrement in CAT activity	Li et al. (2009)
	50 µM	Mung bean	7 day	Abnormality in root and shoot length, root/shoot ratio SOD, GPX and GR activity increased but APX activity decreased	Singh et al. (2007)
	100 mM and 40 mg kg^{-1}	Rice	10 and 30 day	Reduced germination percentage, root and shoot length, fresh weight. Initial mitigation occurs by higher activities of SOD, APX, POD and GR	Shri et al. (2009), Sun et al. (2008)
	25 µM, 67 µM and 30 mg kg^{-1}	*Brassica*	4, 14 and 30 day	Inhibit root and shoot elongation, root and shoot dry weight; reduced Chl content. SOD, CAT, APX and GR activities found to be increased	Khan et al. (2009), Shaibur and Kawai (2009), Sinha et al. (2010)
	60 mg kg^{-1}	*Cicer*	90 day	Decreased plant water content and growth rate. Increased activities of CAT, APX and total antioxidant index	Gunes et al. (2009)
	668 µM	Maize	8 day	Inhibit plant root and shoot length. Higher activity of APX only. SOD and CAT did not respond.	Duquesnoy et al. (2010)
	500 µM As	Arabidopsis	7 day	Decreased fresh weight. Increased the level of AsA, DHA, GSH and GSSG	Leterrier et al. (2012)

	50 µM	Ridged Gourd	7 day	Reduced Chlorophyll a and b synthesis, carotenoid content and plant fresh weight. SOD, CAT and DHA level increased but APX and GST decreased	Singh et al. (2013)
	30 mg kg^{-1}	Fenugreek	9 week	Reduced plant height, photosynthetic rate and seed yield. Significant increment in the SOD level helps to cope preliminary stage	Talukdar (2013)
	100 and 500 µM	Water Thyme	4 day	SOD increased to ameliorate stress	Srivastava et al. (2013)

index (Li et al., 2007). However, the germination decreases with the increment in the metal concentration. For example, 100 and 200 μM concentration of As in *Vigna mungo* reduced seed germination around 13–30% (Srivastava & Sharma, 2013). In case of rice, 500 mM As (V) and 50 mM As (III) not only reduced the germination rate, but also greatly inhibited root length, shoot length and fresh weight (Shri et al., 2009). Similarly, treatment of *Phaseolus aureus* with As (1, 25, 50 and 100 mg l^{-1}) was found to decrease the germination rate seed vigor and germination index (13–74%) (Patel et al., 2013).

10.3.2 Plant Growth and Development

Cd and As are not considered as essential components for plant growth and development; hence, their overaccumulation stimulates abnormalities in plants (Zhang et al., 2019; Hasanuzzaman et al., 2015). Since root is the first exposed plant part that directly comes in contact with the heavy metals, they are the most affected of all the plant organs. In rice plants, after exposure to As stress, impaired growth has been observed throughout and highest deposition occurs in root tissue followed by traces in leaf transporters. Necrosis, decomposition, highly mucilaginous and reduced root occur as symptoms of As and Cd-exposed plants. As a result, shoot development is lowered simultaneously causing leaf rolling and chlorosis. It has been found that Cd and As stress reduced lateral root and root hair formation, making the root rigid, brown and twisted (Krantev et al., 2008; Styblo et al., 2000; Naranmandura et al., 2011). On a closer look, it actually inhibited mitotic cell division in the meristematic cells, wherein the root diameter increased only due to higher development of cortical and parenchyma cells. The inhibition of cell division, chromosomal aberrations, impaired mitotic index and micronuclei due to application of exogenous Cd has been well established (Seth et al., 2008).

Changes in leaf phenotypes in terms of higher chlorosis, lower leaf lamina and sometimes wilting occur due to Cd and As toxicity. Treatment of seedlings of rice, wheat and tomato with Cd and rice, mung bean, maize, *Triticum* and *Brassica* with As resulted in desiccation, stunted growth and necrosis, which led to decreased plant shoot and root elongation. Under Cd stress, remarkable reduction in the leaf area and relative water content has been observed, which might be due to lower nutrient uptake, photosynthesis, respiration, water conductivity as well as assimilation of carbon and nitrogen (Rizwan et al., 2016). Plants treated with As-contaminated irrigation water showed major loss of Ca, S, P, K and Mg nutrients and heavy metal accumulation culminating in reduced growth rate of the plants (Sultana et al., 2012). A few literatures on how higher toxicity inhibits plant growth and leads to serious damage have been listed in Table 10.1.

10.3.3 Photosynthetic System

Injury of photosynthetic apparatus is the primary signature of any kind of oxidative stress (Upadhyaya et al., 2021). Cadmium and different forms of As are effective inhibitors of photosystems, as shown by many studies (Vassilev et al., 2005; Stoeva & Bineva, 2003). Toxicity of Cd and As generally targets photosynthetic apparatus like light harvesting complex, Photosystem I and II (Hasan et al., 2009; Singh et al., 2006; Duman et al., 2010). In fact, they reduce the stomatal opening capacity such that these apparatuses do not get proper access. Iron (Fe) plays a crucial role in chlorophyll as well as pigment formation. Cd inhibits iron reductase that reflects in lower photosynthetic yield (Akinola & Ekiyoyo, 2006). As and Cd target biosynthesis process of photosynthesis-related pigments (chlorophyll and carotenoids) and primarily decrease their content in the epidermal cells (stomata and its guard cells) followed by mesophyll cells (Jain & Gadre, 2004; Haider et al., 2021). Therefore, stomatal conductivity, cell division, cell elongation and stomatal density in the epidermal cells were reduced, resulting in lower water conductivity. Efficient photosynthesis and carbon fixation are driven by two vital enzymes, ribulose-1,5-bisphosphate carboxylase/oxygenase (RuBisCo) and phosphoenolpyruvate carboxylase (PEPCase) (Zhang et al., 2007). RuBisCo activity can be inhibited by Cd toxicity and it can also replace Mg ions that are essential for carboxylation reactions. As a result, RuBisCo move toward oxygenation reaction or photorespiration, which leads to the loss of carbon (Tran & Popova, 2013). As-stress reduces photosynthetic electron transport in the thylakoid membrane. The process of ATP and reduced nicotinamide adenine dinucleotide phosphate (NADPH) production in the carbon fixation reactions center is reduced by As. During photophosphorylation, Pi can be replaced by As(V), which causes disbalance in the electron transport chain and ATP synthesis (Avron & Jagendorf, 1959; Watling-Payne & Selwyn, 1974). Notably, in rice plants, the efficiency of RuBisCo has been found to decrease in presence of toxic As level. As a result, the carbon fixation capacity of RuBisCo large subunit as well as photosynthetic efficiency decreased (Ahsan et al., 2010). Following that, the encoded plastid DNA of large subunit was found to suppress the expression level (Bock, 2007). Taken together, the toxic effect of As and Cd drastically reduced the net photosynthetic rate, electron transport flow, Fv/Fm ratio, Chl a/b ratio, PS II efficiency and overall photosynthetic apparatus (Srivastava et al., 2013).

10.3.4 Oxidative Damage

Metals can generate oxidative stress directly by generating ROS or by applying primary stress, resulting in secondary oxidative damage. As stated in Sect. 10.3.3, both As and Cd can inhibit the chloroplast and mitochondrial electron transport system, thus generating detrimental ROS in plant cells. In some plants, namely, pea and rice, Cd treatment stimulates the plasma membrane bound NADP of peroxisome production, which immediately accumulate ROS (Hasan et al., 2009; Tran & Popova, 2013). Among the heavy metals, Cd followed by As are the fastest in generating ROS in plant system. Variable concentration of exogenous Cd treatment results in the overproduction of NADPH-oxidase, accumulation of H_2O_2 in fatty acids, decrement in homoglutathione and glutathione levels and finally results in a redox imbalance (Gill & Tuteja, 2010). Lipid peroxidation is the marker for detection of oxidative stress. Root tissues of several plants displayed higher MDA content and their As accumulation was also directly correlated with H_2O_2 accumulation (Van Breusegem & Dat, 2006; Møller et al., 2007; Shri et al., 2009). ROS-mediated hydroxyl (OH) and superoxide ($O^{\cdot -2}$) radicals can cause direct double-strand DNA breaks (Das et al., 2022). This threatens the mitotically active cells resulting in recombinogenic lesions in chromosomal DNA and chromosomal abnormalities. Some notable damage of DNA hampers cellular development processes, like higher frequency of anomalies, including laggards, precocious separation, fragments, stickiness, and double and single bridges (Kranner & Colville, 2011). A number of oxidative stress-responsive proteins take part in amelioration of ROS-generated hydrogen peroxide and superoxide. Antioxidant enzyme helps to stabilize free radicals before oxidation to cellular component. These include nonenzymatic scavengers like ascorbic acid (ASA), tocopherols, glutathione (GSH), carotenoids and enzymatic scavengers like peroxidase (POD), ascorbate peroxidase (APX), superoxide dismutase (SOD), catalase (CAT), glutathione reductase (GR), dehydroascorbate reductase (DHAR), monodehydroascorbate reductase (MDHAR), etc. (Das & Roychoudhury, 2014). Arsenic detoxification process, e.g., reduction of As(V) to As(III) were also reported to generate ROS (Meharg & Hartley-Whitaker, 2002). To improve the tolerance level, various plants (sunflower, rice, common bean, wheat, coontail and pea) showed significant up-regulation in the antioxidative enzyme level (Dixit et al., 2001; Baruah et al., 2019; Gallego et al., 1996; Shah et al., 2001; Hasanuzzaman & Fujita, 2013; Norton et al., 2008; Abercrombie et al., 2008; Chakrabarty et al., 2009). Table 10.1 enlists literature survey of some of antioxidant enzymes and their role in different plants.

10.3.5 Nutrient Uptake and Water Relation

The existence of As and Cd in soil inhibits absorption of nutrient elements, K, S, Ca, Zn, Mn, and B, etc. by roots. Particularly, Cd can bind with Ca, Mg, P and K, limiting their translocation from soil to root due to Cd-mediated complex formation (Tran & Popova, 2013). In barley, treatment of soil with Cd was found to generate nutrient (P, K, Ca, Mg, Cu, Fe, Mn, Zn, Mo and B) deficit plants with minimized growth pattern (Gou et al., 2007). Upon uptake of excessive heavy metals like As and Cd, the cell-cell sodium and potassium channels were impaired. It also decreased the uptake of K and Ca. In particular, As and Cd inhibits rhizobia infection and root nodule formation by lowering the assimilation of ammonia (Kinay, 2018). Later studies found out that As (III) caused disruption of gene expression involved in early nodule development (Lafuente et al., 2010). In fact, it can prevent the absorption of nitrate, thereby reducing the nitrate reductase enzymatic activity (Tran & Popova, 2013). Cd was found to affect the chelation of iron and subsequently its pump toward xylem, resulting in unhealthy cellular development (Popova et al., 2012). In case of As(V) exposure, supply of inorganic N to the assimilation pathway impacted and transporter required for NO^{-3} and NH^{+4} transportation were down-regulated (Norton et al., 2008; Chakrabarty et al., 2009). Moreover, transcript of nitrate reductase was down-regulated significantly (Chakrabarty et al., 2009). Taken together, As and Cd toxicity led to a massive blow to the entire amino acid pool and total protein content along with typical reduction in the metabolic processes.

During heavy metal stress, osmotic stress or water imbalance is most common among the generated secondary stress (Rucińska-Sobkowiak, 2016). It was found that plant organs (i.e., leaves, roots and shoots) faced water deficiency due to exposure to As and Cd in variable plant species including bladder campion, clover, alfalfa, rice, common bean, white lupin, rice, wheat, lettuce, sunflower, barley, pea, etc. (Schat et al., 1997, Rucińska-Sobkowiak, 2016; Stoeva et al., 2005; Hasanuzzaman et al., 2015). Metal stress majorly affects the root region by decreasing root hair surface, enhanced root dieback, primary root elongation, and finally abnormal root development (Lux et al., 2011; Hasanuzzaman et al., 2015). The main reason demonstrated by several groups is that the osmotic ability of root sap turns higher than Cd-enriched soil, which limits the rate of water absorption for root tissue and subsequent generation of dehydration stress (Małecka et al., 2008). Metal ions can also increase the cell wall thickening that inhibits the apoplastic water flow (Hashem, 2014). Similarly, diffusion of Cd and As significantly suppresses the aquaporin functionality and membrane protein integrity (Gall et al., 2015). All these effects further culminate in decreased water conductivity in soil to root and subsequent root to other organs. In addition, overaccumulation of such metals in membrane and cell wall region lower the release of exudates from root channel (Kaznina et al., 2014). In response to As or Cd stress, plants synthesize more osmoprotectants like sugar, proline and glycine betaine to maintain the Na channel, water balance and cellular homeostasis (Xu et al., 2010; Garg & Singla, 2012).

10.4 Detoxification Strategies of Plants to Fight Arsenic and Cadmium Stress

In response to toxic effects of these heavy metals, plants have also developed methods to fight these contaminants with some detoxification strategies. In plants, detoxification of heavy metals (Cd and As) from cell is a complex process. A series of mechanisms initiates within cell to reduce the toxicity of Cd and As. Detoxification can be achieved by applying cell-wall deposition, cytoplasmic chelation, sequestration of these metals into vacuoles (vacuolar sequestration) and inducing their efflux (Zhang et al., 2022; Luo & Zhang, 2021).

10.4.1 Cell Wall Deposition of Cd and As

Cell wall can act as a first barrier against heavy metal contamination. It is generally made up of cellulose, hemicelluloses and pectin. Previous studies showed that, among these cell wall components, pectin is the most important one, which binds with heavy metals, thereby preventing its transport to the cytoplasm (Haynes and Haynes, 1980; Grignon & Sentenac, 1991). Recent studies showed that, after entry of Cd into cells, net electronegative charges of cell wall hold cations to the cell wall, thereby protecting cell from damage and preventing its transport to cytoplasm (Peng et al., 2017). Previous studies with *Populus euphratica* showed that the *PeXTH* (xyloglucan endotransglucosylase) gene is present in the cell wall and involved in Cd tolerance. Overexpression of this gene reduced the Cd accumulation by reducing the Cd binding sites in cell wall, thereby alleviating Cd toxicity (Han et al., 2014). Xyloglucan provides sites for Cd binding in cell wall and overexpression of *XTH* gene promotes degradation of xyloglucan on the cell wall, which is crucial for Cd binding. Therefore, less amount of xyloglucan in the cell wall led to a decrease in Cd binding sites, thereby reducing overall Cd accumulation and transport into the cell. Belleghem et al. (2007) showed that, when Cd contamination is low in soil, it is stored in the cell wall. Later, Yu et al. (2020) showed that Cd deposition in cell wall of root is an effective strategy for developing Cd-safe rice lines. In rice, *OsCDT1* plays an important role for increasing tolerance to Cd. *OsCDT1* can chelate Cd on the cell surface, thereby limiting Cd to enter into the cell (Liu et al., 2014). Although not many studies have been reported for As-stress, recently a report by Huang et al. (2021) showed that roots of cabbage plants showed changes in root cell wall matrix in response to As. Studies also showed that exocellular sequestration can potentially serve as a mechanism of As detoxification in *Pteris vittata* (Datta et al., 2017).

10.4.2 Cytoplasmic Chelation

Cytoplasmic chelation of heavy metals in cells can act as a second line of defense against Cd and As contamination. In cytoplasm, glutathione (GSH) and phytochelatins are involved in chelation process of metals (Grill et al., 1985, 1989). It is also important to maintain the cellular redox balance. Glutathione or phytochelatin formed complexes with heavy metals. In case of GSH, studies showed that it forms complexes with Cd (Cd-GS$_2$), and is then transported to the vacuoles, reducing the cytotoxicity of Cd. In yeast, YCF1 (Yeast Cadmium factor 1) transporter is able to transport this complex into the vacuoles (Li et al., 1997), thus lowering the Cd concentration in cytoplasm. Other than GSH, phytochelatins are small cysteine-rich peptides, synthesized from GSH by the activity of enzyme PCS (phytochelatin synthase). Clemens et al. (1999) showed that in case of wheat (*Triticum aestivum*), a gene (*TaPCS1*) is important for phytochelatin synthesis. TaPCS1 (a 55 KDa protein) and its activity is important for providing Cd tolerance. Furthermore, studies showed that *TaPCS1* gene expression was suppressed when the plant was treated with the inhibitor of GSH biosynthesis. Similarly, disruption of *PCS* gene in *Schizosaccharomyces pombe* showed hypersensitivity to Cd and cells were unable to produce phytochelatins (Clemens et al., 1999). Parallel to these reports, Yurekli and Kucukbay (2003) showed that in sunflower (*Helianthus annunus*), production of phytochelatins enhanced Cd tolerance. Recently, it is believed that PC-mediated Cd and As detoxification process involves formation of complexes with heavy metals. In cytoplasm, PCs formed complexes with metals and metalloids and finally transported to the vacuoles. The PC-heavy metal complexes are then transported to the vacuoles by the activity of the ABC transporters. In case of rice, ABCG36 and ABCG43 function is important for transportation of PC-Cd or PC-As complexes (Fu et al., 2019; Oda et al., 2011). Similarly, in *Arabidopsis*, the functions of ABCC1 and ABCC2 are associated with Cd detoxification (Song et al., 2014). Schmoger et al. (2000) showed that As detoxification in cell suspension of *Rauvolfia serpentina* and seedling of *Arabidopsis* is mediated via metal-binding phytochelatins.

10.4.3 Vacuolar Sequestration

In case of vacuolar sequestration, detoxification of these contaminants from the cytoplasm to the vacuole is favored by the activity of the vacuolar membrane localized transporters. Activity of these transporters further helps in compartmentalization of these metals into vacuoles. Based on the reports in plants, few members of HMA, ABCC, CAX and NRAMP family are localized in the vacuolar membrane and their activity is important for alleviating metal toxicity in plants (Pottier et al., 2015; Morel et al., 2009; Chao et al., 2012). HMA is a member of the heavy metal ATPase family, which is also known as P1B-ATPase (Mills et al., 2003; Mills et al., 2005; Miyadate et al., 2011). Its function is associated with ATP hydrolysis and metal ion transport across the membrane. It contains E1-E2 ATPase domain and haloacid dehalogenase like hydrolase domain. In case of *Oryza sativa*, *OsHMA3* localized in the vacuolar membrane is mainly expressed in roots and involved in Cd transport from cytoplasm to the vacuole (Ueno et al., 2010; Miyadate et al., 2011). Therefore, it prevents the loading of Cd in the root xylem tissues. Ueno et al. (2010) reported in case of *Oryza sativa japonica* cv. Nipponbare that knockdown of *OsHMA3* by RNAi mediated silencing can increase the overall shoot Cd accumulation. Overexpression of *OsHMA3* gene can reduce the shoot Cd accumulation. Along with loss-of-function of this gene, a single amino acid substitution showed remarkably high accumulation of Cd in shoots and grains. This is also similar in case of other HMA3 members of *Arabidopsis*, *Nicotiana*, *Sedum alfredii* and *Sedum plumbizinicicola* plants. In case of Arabidopsis, Morel et al. (2009) showed that *AtHMA3* gene function is important for vacuolar sequestration of Cd. Similarly, a study by Liu et al., (2017) reported that *SpHMA3* expression is important for detoxification of Cd by facilitating vacuolar sequestration. Similar to the above reports, Ueno et al. (2011) reported that *Thlaspi caerulescens TcHMA3* is a vacuolar localized transporter and its activity is highly specific for Cd. Recently, a study of Shao et al. (2018) showed that expression of *OsHMA3* under the regulation of *OsHMA2* gene promoter can induce the vacuolar sequestration process of Cd. Being more effective, it sequesters more Cd into vacuoles; thus, this process is helpful to minimize the overall amount of Cd in rice grains.

HMT1 transporter transports Cd complexes into the vacuole. Transgenic approach showed that overexpression of *SpHMT1* can significantly increase the tolerance to Cd in yeast (Prévéral et al., 2009). Similarly, Cd-PC (cadmium-phytochelatin) complexes can be sequestrated into vacuoles by the function of ABCC type of transporters. In case of *Arabidopsis*, AtABCC1, AtABCC2 and AtABCC3 are involved in transferring Cd-PC chelates from cytosol to the vacuoles (Park et al., 2012; Brunetti et al., 2015). Further, functional studies in this regard showed that knockout of these genes caused hypersensitive response to Cd in *Arabidopsis*. Other than HMA, another important detoxification pathway is present in rice. It is an ABC-type transporter family member (OsABCG43) and plays important role for the sequestration of Cd into vacuoles, thereby reducing overall cytoplasmic concentration of Cd (Oda et al., 2011). Similarly, another report concerning CAX family showed that, in case of *Arabidopsis halleri* (Cd hyperaccumulator), vacuolar sequestration of Cd can be achieved by increased expression of *AhCAX1* gene (Baliardini et al., 2015).

Similar to Cd, As is also sequestered into the vacuole as As(III)-PC complexes by the activity of ABC type C transporter. Previous reports showed that in case of rice, *OsABCC1* helps in sequestration of As(III)-PC complexes into the vacuole (Song et al., 2014), thereby reducing the cytosolic concentration of As(III) and minimizing the cytotoxic effects of As. *OsABCC1* is generally expressed in the exodermis and pericycle region of root and its function facilitates vacuolar sequestration. As-PC complex formation with As(III) is important for vacuolar sequestration; therefore, sequestration of As(III) can depend on the overall biosynthesis of PCs in cell. In case of *Oryza*, *OsPCS1* (a phytochelatin synthase gene) function is therefore important, whereas mutant rice with *Ospcs1* gene showed an increased accumulation of As in the rice grain (Hayashi et al., 2017).

10.4.4 Metallothionein-Based Detoxification

Metallothioneins (MTs) are small, metal-ion-binding, cysteine-rich proteins (~45–85 amino acids) in cells. They are characterized by high percentage of cysteine residues with multiple roles in plants. Previous studies showed that plant metallothionein plays an important role in providing tolerance to heavy metal stresses, especially for copper (Cu) and cadmium (Cd). Metal binding activities were studied with different MTs isolated from different plants. In Cd(II) containing medium, *Quercus suber, QsMT1* contains six Cd(II) ions and *QsMT2* contains four Cd(II) ions (Freisinger, 2009; Domènech et al., 2007). In case of *Cicer arietinum*, *CaMT2* contains five Cd(II) ions. Zhou and Goldsbrough (1994) and Murphy and Taiz (1995) showed that *MT* genes are induced by Cd and Zn stress. Similarly, plant *MT* genes when expressed in metal-sensitive yeast, are able to restore Cu, Cd and Zn tolerance. Hassinen et al. (2009) showed that in hybrid aspen (*Populus tremula* × *tremuloides*) grown at metal-contaminated site, *PttMT2b* expression was induced with Cd and Zn concentrations. In case of *Arabidopsis*, the function of MT1 family member is important for Cd and Zn tolerance (Zimeri et al., 2005). Lee et al. (2004) showed that *AtMT2a* and *AtMT3* gene function is important for providing Cd tolerance. In case of *Brassica juncea, BjMT2* function is associated with Cd and Cu tolerance (Zhigang et al., 2006). Similarly, a MT1 family member from *Cajanus cajan* (*CcMT1)* confers enhanced tolerance to Cd and Cu in *Arabidopsis* and *Escherichia coli* (Sekhar et al., 2011).

10.4.5 Plant Defensins

Plant defensins are another class of cysteine-rich molecules that play an important role in Cd detoxification by chelation. Few members of this family are effectively involved in Cd tolerance. Localized in the cell wall of root xylem parenchyma cell, they chelate cadmium forming complexes which are then moved to the apoplast to minimize the cytotoxic effects of heavy metals. Luo et al. (2019b) showed that in *Arabidopsis*, AtPDF2.5 is involved in detoxification of cytoplasmic Cd. Similarly, later reports suggested that AtPDF2.6 function is also similar to AtPDF2.5 (Luo et al. 2019a). Therefore, loss of function of AtPDF2.5 and AtPDF2.6 reduced the overall Cd tolerance. Based on the studies, heterologous expression of this gene in *Escherichia coli* and yeast also provided Cd tolerance. Furthermore, studies identified a portion of eight cysteine residues as important in PDFs, which is most important for chelation (Luo et al., 2019a, 2019b). In *Brassica*, BnPDFL function was also similar to plant defensins and played a positive role in Cd stress. In rice, CAL1 is a QTL (Quantatative Trait Loci) that encodes a defensin-like protein. Activity of CAL1 (Os02g629800) regulates Cd translocation process from root to shoot (Luo et al., 2018).

10.4.6 Cd and As Efflux

In rice, activity of several plasma membrane associated transporters directly excludes heavy metals out of the cell. Therefore, activity of these transporters is considered important and expression of these members effectively minimizes the cytotoxic effects of heavy metals like Cd and As. Previous studies showed that OsHMA9 (a P-type heavy metal ATPase) located in the plasma membrane is important for efflux of Cd, Cu, Zn and others. Knockout of *OsHMA9* gene accumulates greater amount of Cd in cells as compared to the wild type (Lee et al., 2007). Similarly, OsZIP1 can act like a metal efflux transporter. Activity of OsZIP1 is important for efflux of Cd, Zn and Cu (Liu et al., 2019). Furthermore, advances in research also identified that a G-type ATP binding transporter activity is also important for efflux of heavy metals. In rice, OsABCG36/OsPDR9 is involved in Cd tolerance (Fu et al., 2019). Generally, it is expressed in the root zones and helps to maintain a lower Cd content in root cells. Apart from Cd, Xu et al. (2017a) showed that OsHAC4 plays an important role in As detoxification from cell. Activity of *OsHAC4* gene in root tissue facilitates the efflux of As(III), minimizing the cytotoxic effects of As(III).

10.5 Bioremediation Techniques to Alleviate Arsenic and Cadmium Stress

Two different strategies can be applied in agriculture to reduce Cd and As accumulation in rice grains. The first process is to minimize the bioavailability of contaminants from soil and the second process is to check their uptake and translocation to the grains or seeds. The above amendments can be achieved by changing the soil composition, applying NPs in soil or plants and using genetic engineering approaches, etc. (Zhao & Wang, 2020).

10.5.1 Soil Composition Changes

For minimizing the bioavailability of heavy metals in soil, several soil amendments are being performed in agriculture that effectively limit the uptake and accumulation of heavy metals in plant. It is considered that soil acidity (low pH in soil) is the major reason behind a high uptake of Cd in rice. Therefore, liming can be performed in agriculture to increase the soil pH. Chen et al. (2018) showed that liming is a highly effective strategy that limits the Cd accumulation in rice grain. Application of 7.5 t ha^{-1} of $CaCO_3$ in land before rice planting raised the pH of the soil from 5.5 to 6.5, thereby reducing the grain Cd concentration nearly 70–80%. Similar to this study, application of biochar in rice also lowered the accumulation of Cd to the rice grains (Bian et al., 2013, 2014). Other than Cd, application of iron oxides/hydroxides and manganese oxides in agricultural land can facilitate immobilization of arsenic, thus limiting its uptake in the cell. Grain As concentration can be effectively lowered by the application of Fe-rich materials and manganese oxides. Generally, manganese oxides promote the oxidation of As(III) to produce As(V), thereby increasing the absorption of As in solid phase (Honma et al., 2016; Xu et al., 2017a, 2017b; Yu et al., 2017).

10.5.1.1 Biochar and Nutrient Management

In recent years, addition of biochar in heavy metal contaminated soil has provided much improvement in agriculture (Zhang et al., 2019; Liang et al., 2021). Biochar is generally produced in limited oxygen condition (slow-pyrolysis) and contains organic manure and crop residues (Ding et al., 2016). Studies with biochar showed that incorporation of biochar in agricultural land effectively reduced the Cd availability and accumulation, thereby reducing overall Cd toxicity and its translocation to grain. Different types of biochar parameters (pyrolysis temperature, feedstock type) and physiochemical properties (pore space; surface area) are used in biochar preparation, which is important for increasing the absorption ability of heavy metals. Besides, different types of absorption mechanisms, namely, mineral absorption, electrostatic attraction, cation interaction and ion exchange are associated to improve absorption ability of biochar. Puga et al. (2015) showed that sugarcane (*Saccharum officinarum*)-derived biochar can significantly decrease Cd uptake process in plants. Similarly, Rizwan et al. (2018) showed that biochar significantly reduced the effect of Cd toxicity on rice. Qiao et al. (2018) showed that the application of biochar reduced the uptake of Cd in rice roots. Same kind of result was also shown by Bashir et al. (2018) and application of rice straw biochar and rock phosphate addition increased immobilization of Cd in soil. In wheat, biochar application increased the growth and yield in Cd contaminated soil (Abbas et al., 2017, 2018). Biochar in general raised the pH of the soil, which played most important role for Cd immobilization. Other than Cd, Wu et al. (2018) showed that iron modified biochar is beneficial for providing remediation of arsenic (As)-contaminated paddy soil. Previously, Beesley et al. (2013) showed that application of biochar in soil effectively reduces the uptake of As in tomato (*Solanum lycopersicum*) than their respective control plants. Tomato plants grown in biochar-applied soil accumulated less amount of As in root and shoot and the least in fruit.

Other than the biochar, application of nutrients is also important/beneficial for minimizing the toxicity of Cd in plants. Sufficient and exact amount of nutrients, if applied at proper time, can mitigate the toxicity of Cd and strengthen plant defense (Qin et al., 2020). However, excessive application of nutrients can increase soil acidity, thereby increasing the solubility of Cd and also bioavailability. Ning et al. (2017) showed that application of high amount of nitrogen and other nutrients at high doses increase soil pH, increasing the solubility and bioavailability of Cd. Though an excessive amount of nutrients induces the uptake of heavy metals, studies reported that a few nutrients can effectively minimize the damage induced by Cd stress. Likewise, application of selenium at low concentration can improve the overall plant performance under Cd stress. Reports showed that application of 2 μM sodium selenate can effectively minimize the impact of 600 μM cadmium in soil (Nazar et al., 2012). Application of Se minimizes lipid peroxidation process, and improves the function of chlorophyll, maintains a high activity of antioxidant enzyme and maintains better integrity of membrane (Zembala et al., 2010; Wu et al., 2017). Along with Se, Silicon (Si) also serves the same by improving plant growth and photosynthesis. Nazar et al. (2012) showed that application of 0.6 μM of Si can improve the performance of plants treated with 2.5 μM Cd. Si-applied plants effectively maintain the overall chlorophyll content and chlorophyll fluorescence parameters related to photochemical quenching. Later, Feng Shao et al. (2017) showed that application of Si down-regulated the expression of key transporters that are associated with Cd uptake in the cell. Furthermore, studies with Si showed that Si down-regulated the expression of OsNRAMP5 and OsHMA2, thereby minimizing Cd uptake in roots and accumulation of Cd to the upper portion of the plant.

10.5.2 Application of Nanotechnology

In the last decades, nanotechnology has been playing important roles for increasing the agricultural productivity. Nanotechnology is providing remedies to different types of stresses including heavy metal stress. Different types of nanoparticles (NPs) are being used in agriculture, which helps in minimizing the bioavailability of Cd in soil or its uptake. Previous report showed that application of Fe_3O_4-NPs reduced Cd mobility in soil (Sebastian et al., 2019). Besides, application of nanoparticles sometimes increased the soil pH, preventing its transport into root. Likewise, hydroxyapatite NPs can release phosphate in soil and increase the soil pH, reducing the harmful effects of heavy metals and minimizing its uptake (Cui et al., 2018).

Other than the increase of soil pH, formation of apoplastic barriers in plant roots can be induced by the application of NPs. This apoplastic barrier may reduce the transportation of heavy metals into roots by creating barriers for the movement of heavy metals in cells (Rossi et al., 2017). In general, different types of NPs like cerium dioxide nanoparticles (CeO_2-NPs) and titanium dioxide (TiO_2-NPs) used in agriculture lowers the bioavailability of Cd and As or reduces their accumulation in the essential portion of cell. Previous studies showed that in rice, application of CeO_2-NPs (200 mg/L) in the foliar portion can effectively induce the ROS scavenging activities (Wang et al., 2019), minimizing the oxidative stress related cellular damage. Similarly, in *Brassica napus*, application of CeO_2-NPs improved the photosynthetic performance under Cd stress. Based on the results, net efficiency of PS-II (F_v/F_m) was maintained in CeO_2-NP treated plants (Rossi et al., 2019). Previous studies by Rossi et al. (2016) showed that CeO_2-NP application overall improved the uptake of Mg^{2+} ion. Magnesium is an essential element of chlorophyll and might be involved in better maintenance of photosynthesis.

Apart from CeO_2-NPs, Wang et al. (2010) showed that application of TiO_2-NPs also improved the activity of the antioxidant enzymes in corn (*Zea mays*). Similarly, Singh et al. (2016) reported that in case of soybean (*Glycine max*), application of TiO_2-NPs improved the photosynthetic rate and limited the toxicity of Cd. In case of rice, TiO_2-NPs are important to maintain the growth, root length, and a high activity of ROS scavenging enzymes, thereby alleviating the damage caused by Cd. In maize, foliar application of TiO_2-NPs helped to maintain growth and biomass under Cd contaminated soil. Other than these, iron oxide and zinc oxide NPs improved the overall performance of the plant. Ali et al. (2019) showed that combined application of ZnO-NPs (zinc oxide nanoparticles) and biochar (in soil) effectively minimized the cytotoxicity of Cd. In case of rice, application of ZnO-NPs also lowered the level of As in the shoot and root tissues (Yan et al., 2021).

10.5.3 Genetic Engineering Approach

Genetic engineering/gene editing techniques can be used in plants to minimize the cytotoxic effects of heavy metals. In general, either knockout or overexpression of a specific gene limits the uptake or transportation of heavy metals in plants. In rice and other plants, several gene family members are actively involved in uptake and transportation of heavy metals. Likewise, NRAMP gene family members (*NRAMP1* and *NRAMP5*) and IRT gene family (*IRT1* and *IRT2*) members help in uptake of Cd. HMA family member (*OsHMA2*), *OsLCT1* and other members are involved in transportation of Cd. To reduce the cytotoxic effects of heavy metals, researchers are targeting these transporters that help in uptake, transportation and efflux.

In rice, OsNRAMP5 is a major transporter of Cd uptake. Tang et al. (2017) showed that knockdown of *OsNRAMP5* via CRISPR/Cas9 gene editing technique limited the overall Cd accumulation in rice grain. Modified lines showed 90% reduction in grain Cd concentration when grown on Cd contaminated soil. Similarly, OsLCT1 is another important gene in rice, involved in Cd transport. Knockout of *OsLCT1* by RNAi silencing reduced the overall accumulation of Cd in rice grain (Uraguchi et al., 2011). Similarly, previous reports showed that OsHMA2 is a plasma membrane associated transporter involved in root to shoot translocation of Cd and Zn (Satoh-Nagasawa et al., 2012; Takahashi et al., 2012; Yamaji et al., 2013). Disruption *OsHMA2* effectively reduced the transportation and distribution of Cd to the panicle and flag leaf. OsZIP7 is another plasma membrane associated member located in the pericycle region of root and facilitates the xylem loading of Cd and Zn (Tan et al., 2019). Knockout of *OsZIP7* also lowered the root to shoot translocation of Cd. OsCCX2 is a putative cation/Ca exchanger, expressed in the xylem region and facilitates transportation of Cd and Ca (Hao et al., 2018). Knockout of *OsCCX2* gene lowers the overall Cd transport in rice grain.

Similarly, transgenic rice overexpressing the *OsABCC1* could trap more As into vacuoles, reducing the overall accumulation of As in grain (Deng et al., 2018). As(III) sequestration in vacuoles can be improved by overexpression of *Pteris vittata* (*PvACR3.1*) gene (Chen et al., 2019). Transgenic rice overexpressing *PvACR3.1* could lower the overall accumulation of As into rice grains (~26–46%). In *Arabidopsis*, overexpression of *PvACR3.1* enhanced As translocation in shoots, allowing the development of As hyperaccumulator plants. Other than these, aquaporins are also helpful for As detoxification process. Overexpression of NIP family members like *OsNIP1.1* and *OsNIP3.3* effectively reduced the grain As concentration (Sun et al., 2018). Duan et al. (2015) showed that two inositol transporters (*AtINT2* and *AtINT4*) are associated with arsenite uptake in *Arabidopsis*. Disruption of these gene members effectively lowered the As concentrations in phloem. Arsenic can be transported into cell through the phosphate transporters. In *Arabidopsis thaliana* (*AtPHT1.1/4*) (Shin et al., 2004), *Pteris vittata* (*PvPHT1.3*) (DiTusa et al., 2016) and *Oryza sativa* (*OsPHT1.8*) (Wang et al., 2016) members of phosphate transporters are present in cell, which facilitate the transportation of As(V). Wang et al. (2016) also showed that knockdown of *OsPHT1.8* lowered As(V) uptake by 33–57%, thus improving the As(V) tolerance. Similarly, in Arabidopsis (ARQ1-arsenate reductase QTL1) (Sánchez-Bermejo et al., 2014) and rice (HAC1- high arsenic content1) (Chao et al., 2014) As(V) tolerance was provided by the respective genes. HAC1 reduces As(V) to form As(III) at the outer portion of root and efflux out As to the outer environment (Chao et al., 2014). Shi et al. (2016) showed that overexpression of *OsHAC1.1* and *OsHAC1.2* effectively minimized the As content in rice grain (nearly 20% lesser As).

10.6 Conclusion and Future Perspectives for Metal-Free Agriculture

Heavy metal contamination in agricultural practice is an extremely worrying factor that needs to be resolved soon for safe consumption of plants by humans and animals. Common strategies are unable to completely render the global agriculture free from heavy metal contamination. Among these heavy metals, As and Cd toxicity are one of the topmost concerns. Their uptake mechanism by the plant roots, transport through the cortical cells for xylem- and phloem-loading have been well-studied, and increasingly so. The number of transporters involved is also increasing day-by-day, and hence identification of new transport routes in the near future will open up more ways to block their uptake, transfer and accumulation in plant parts. At the cellular level, the role of plasma membrane transporters in influx/efflux, chelation with phytochelatins and vacuolar sequestration has been studied. These studies open up future research avenues to identify the molecular basis of chelation and how these genes could be utilized for developing heavy metal-tolerant plants. Moreover, among the many methods implemented to encounter heavy metal stress, use of NPs is a fascinating approach that is still in its infant stage. Therefore, how these NPs function at the molecular level to bring about detoxification of As and Cd stress needs to be worked out. Moreover, genetic engineering approaches with novel genes should be further exploited to efficiently deal with As and Cd toxicity. Available literatures have thrown ample light on heavy metal contamination related impact on plants and many remediation approaches are also being increasing attempted. Hence, with further studies being conducted and remediation approaches being scientifically dissected out, the hope to make the global agricultural practice free from heavy metal contamination does not seem too far.

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Chapter 11
Remediation of Persistent Organic Pollutants Using Advanced Techniques

Amtul Bari Tabinda, Rimsha Javed, Abdullah Yasar, Adeel Mahmood, and Rizwan Rasheed

Abstract Persistent organic pollutants (POPs) have adverse effects on the environment as they accumulate in the fatty acids and have a half-life on the scale of years to decades. These pollutants are contaminants of emerging concern (CECs) owing to their biological concerns related to health impacts and also economic benefits. In the US, UK, and Asia about 20% of the food is reported to be contaminated with low levels of POPs and there is no scientific accord on whether these low-level POPs are a matter of concern for humans. The treatment and removal of these organic pollutants are difficult as they travel to far areas through different mediums such as food, air, water, and soil. Remediation of POPs from air, water, soil, and other environmental compartments is crucial and a fundamental concern. For the remediation of POPs conventional treatments such as physio-chemical methods, i.e., coagulation, flocculation, oxidation, and adsorption have been utilized extensively in past decades. However, one of the useful methods of remediating POP pollution in the environment is bioremediation. Biological methods are expected to be a tremendous accomplishment in the cleanup of POPs by having advantages over conventional process. There are different methods that are used to remove these pollutants such as bioremediation, osmosis, adsorption process, membrane technology, and AOPs. For the implementation of these techniques there is a need to consider other factors such as economy, potential, and technical feasibility. Different research projects have been carried out and attempts have been made to find alternatives that are more cost effective. POPs are of great importance for some industries; thus, they are not totally avoidable. The only solution to this problem is to adapt techniques that can degrade the POPs effectively.

A. B. Tabinda (✉) · R. Javed · A. Yasar · R. Rasheed
Sustainable Development Study Centre, GC University, Lahore, Pakistan
e-mail: amtulbaritabinda@gcu.edu.pk

A. Mahmood
Department of Environmental Science, GC Women University, Sialkot, Pakistan

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11.1 Introduction

The class of pollutants that of a nonbiodegradable nature and accumulate in the environment are called persistent organic pollutants (POPs). These pollutants are used in cosmetics, industrial chemicals, pesticides, and are the byproducts of many reactions. POPs have adverse effects on the environment as they accumulate in the fatty acids and have half-life on the scale of years to decades. Half-lives of different polychlorinated biphenyls (PCBs) range from 5 to 20 years and polychlorinated dibenzo-p-dioxins and furans have a half-life of 10–100 years. Length, persistence, long-range transport, and the lipophilic nature of POPs makes them highly toxic at low concentrations. The toxic, persistent, lipophilic, and carcinogenic nature of POPs make them of greater concern.

Different environmental conditions, for example, temperature causes these pollutants to change into a gaseous form. As a result these pollutants travel to distant areas and are found in places where they are not even used. The hydrolyzation and degradation of these compounds are difficult owing to carbon halogen bonding, which makes them persistent in the environment. These pollutants transport in the food web as they bio-accumulate in living organisms. These pollutants are contaminants of emerging concern (CECs) owing to biological concerns related to their health impacts and also economic benefits. The organochlorine pesticides, PCBs, polybrominated diphenyl ethers, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated naphthalene are included in POPs. When living organisms are exposed to these pollutants, they cause different health issues such as neurological, immunological, and reproductive disorders.

The origin of these organic compounds (POPs) is anthropogenic and in the past century these pollutants were released in high amounts owing to development in the chemical and agrochemical industries. As these pollutants are persistent and mobile in the environment, they represent long-term problems. Pesticides and PCBs are examples of intentionally produced POPs. Dioxins and furans are the by-product of industrial processes that are produced unintentionally. An excessive number of pollutants has been released into the environment over the last few decades and are not properly managed. These cause massive contamination in the environment.

Through documentation it is clear that POPs biomagnify after they accumulate in the tissues of animals and humans. It causes has a major impact on both human health and on the environment. In the US, UK, and Asia about 20% of the food is contaminated with low levels of POPs and there is no scientific accord on whether these low-level POPs are a matter of concern for humans. Globally, in humans, the level of POPs has declined but those people who are living in localized areas and rely on marine mammals for food and survival are still at a higher risk. There is a need to gather more information and to identify the risk of these low level POPs.

The treatment and removal of these organic pollutants is difficult as they travel to far areas through different mediums such as food, air, water, and soil. There are different methods that are used to remove these pollutants such as bioremediation, osmosis, adsorption process, membrane technology, and AOPs. For the implementation of these techniques there is a need to consider other factors such as economy, potential, and technical feasibility. The removal of these organic pollutants can be carried out by the implementation of bioremediation and by biodegradation. It is reported that there are various enzymes from different microorganisms that are useful for breaking down the complex compound PCBs.

11.2 Remediation Techniques for POP Pollution

The substantial increase in industrialization, extensive anthropogenic activities, and poorly managed discharges initiated POP in terrestrial and aquatic ecosystems that is a significant threat to human health and the environment. The conventional physio-chemical and biological methods to remediate POPs are described in the following sections.

11.3 Chemical Treatment Methods

The physio-chemical treatment techniques work on the principle of mass transition in the water system such as coagulation and flocculation, which are carried out by sedimentation and filtration processes to eradicate contaminants. Temperature, pH, retention time, velocity gradient and pollutant characteristics are dependent factors for removal efficiency of POPs (Karthigadevi et al., 2021; Sun et al., 2020).

The prime limitation of these traditional physio-chemical techniques, including high resistance to POP removal, is due to inadequate interaction. For this some advanced methods have been documented by Prasad (2021) that are effective in POP remediation. These include adsorption, advanced oxidation processes, membrane technology, i.e., reverse osmosis and one of the most attractive options of bioremediation owing to sustainable, versatile, and simultaneous degradation of POPs (Rashed, 2013; Adithya et al., 2021).

11.4 Adsorption

The adsorption process is well recognized for POP elimination depending upon the types of adsorbent and relative doses, pollutant concentration, and solution pH that enhance efficiency. Adsorption is considered a suitable option because of simple design, low investment cost and least toxic by products (Nguyen et al., 2020; Trojanowicz et al., 2019). Clay minerals, activated carbon and other biomaterials that have a greater surface area and porosity are effective at degrading herbicides, dyes, heavy metals, and phenolic waste. The major adsorbents are carbonaceous materials powdered activated carbon, granular activated carbon, carbon filters, carbon cloth, and black carbon for the removal of pesticides. However, they are effective at POP removal but the major drawback is toxic sludge generation, the requirement of maintenance costs, and fouling of the adsorbent (Zhi et al., 2020; Chen et al., 2021; Prasad, 2021). For the treatment of herbicides, cyclodextrin-based polymer, a porous polymeric adsorbent, was investigated and successfully removed bromopropylate pesticide from wastewater (Rashed, 2013).

11.5 Membrane Technology

Membrane technology is a type of physical method that prevents waste materials by providing a barrier to the pollutants, transporting the mass with hydraulic pressure. No additional additive is required. Simply retention of particles, largely dependent on pore size of the membranes, particle size, and affinity to their molar masses. Different types of membrane technology include reverse osmosis, ultrafiltration, microfiltration, and nanofiltration (Gaur et al., 2018; Kumari et al., 2020; Prasad, 2021; Alharbi et al., 2018).

11.6 Advanced Oxidation Processes

In advanced oxidation processes (AOPs) $OH^{\cdot}$ radicals react with contaminants in the water system under specific conditions and the end product is less toxic than initially (Adhithya et al., 2021). The factors that limit the working efficiency are turbidity, reaction time between the OH radical and pollutant and the pH of the water. Sometimes hazardous chemical production takes place owing to oxidative chemical species (Prasad, 2021; Zhi et al., 2020). Ozone coupled with H_2O_2 are strong oxidizing agents that produce OH radical. In Fenton's reaction Fe+2 reacts with H_2O_2 to form an OH radical. This is highly limited owing to selective oxidizing species, the complex reaction, and the costly operation (Musteret & Teodosiu, 2007; Kumari et al., 2020) (Fig. 11.1).

11.7 Bioremediation

The term bioremediation, which includes phytoremediation and rhizoremediation, is an excellent and useful cleanup strategy for soils contaminated with POPs. It is advantageous over other techniques owing to the low cost and preservation of soil

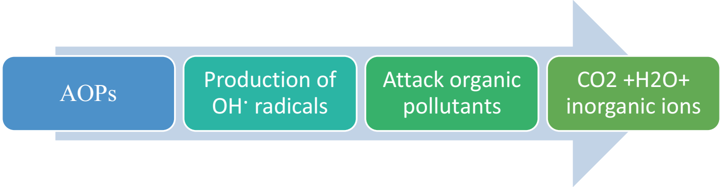

Fig. 11.1 Schematic representation of AOPs working

conditions for better growth of the plants as maintenance of soil function is expected to be a key pillar in the agriculture field (Matsumoto et al., 2009; Chaudhry et al., 2005). The threats to the environment arise with the increase in population rate that results in the substantial increase in anthropogenic activities, intensive use of the extensive industrialization of fertilizers, and this leads to further damage to the overall ecosystem. Soil is now the most impacted domain in the overall regime, as it contains many dangerous pollutants such as POPs, which can stay in the environment for longer periods of time and have major consequences for soil, wildlife, human health, and the whole environment. There are a variety of remedial technologies used to clean up these pollutants in the environment, and bioremediation accomplished using phytoremediation and rhizoremediation are tremendous options (Asante-badu et al., 2020). Biodegradation is a rapidly expanding technology that involves using selected living microorganisms to degrade, metabolize, and immobilize undesired compounds such as pesticides, organic pollutants, and hydrocarbons from soil and water in order to improve their quality. Although any microorganism has the ability to eliminate pollutants, only a few specific or engineered microorganisms, such as *Staphylococcus*, *Bacillus*, *Shigella*, *Enterobacter*, *Streptococcus*, *Alcaligenes*, *Escherichia*, *Acinetobacter*, *Klebsiella*, and *Corynebacterium* are widely used in the remediation of organic pollutants, with *Bacillus* spp. being the most widely used. When it comes to removing persistent organic pollutants, bioremediation technology takes into account the following techniques.

- *Aeration system (Bioventing)* installation: This is a method of aerating soil/water to enhance bioremediation and in situ biodegradation of organic pollutants.
- *Bio-stimulation* is the introduction of nutritive media: This entails modifying contaminated media to offer nutrition to soil bacteria by altering pH and adding limiting nutrients to improve the C: N: P ratio.
- *Bioaugmentation* is the introduction of a microbial strain under regulated conditions: It is a method of degrading organic and inorganic pollutants by combining a microbial community with any biocatalyst such as gene and enzyme. With the help of microorganisms, the degradation process could be carried out under anaerobic conditions, which requires the presence of an electron-withdrawing group for the initial reductive attack or aerobic condition, which requires molecular oxygen such as mono- and dioxygenase, which not only act as a terminal electron acceptor but also as a co-substrate. Water, carbon dioxide, and inorganic salts are formed when functional groups or the basic structure of a chemical are

altered or degraded. In addition, degradation of highly chlorinated compounds is done under anaerobic conditions. Furthermore, because it is difficult for highly chlorinated chemicals to decompose under aerobic conditions, they are degraded under anaerobic conditions (Arsaln et al., 2015; Russell, 2005; Zhao et al., 2022).

Algae, bacteria, and fungi in the microbial community have the ability to biotransform persistent organic contaminants, making them harmless. Different enzymes produced by fungi and bacteria work on a wide range of organic contaminants and aid in the breakdown of organic pollutants. Given that industrial-scale bioremediation of soil contaminated with petroleum hydrocarbons has already been demonstrated, bioremediation may be of potential utility for the treatment of soil and sediment polluted with low levels of PCBs. Biodegradation of PCBs can be done to some extent utilizing a variety of bacteria and/or fungi, according to current knowledge. Anaerobic reductive dehalogenation, in which the PCB works as an electron acceptor and hydrogen replaces chlorine, resulting in the compound being less chlorinated, and aerobic disintegration of the biphenyl structure, which is only active on PCBs that are less chlorinated less than five atoms of chlorine. *Acinetobacter*, *Pseudomonas*, *Comamonas*, *Burkholderia*, *Sphingomonas*, *Acidovorax*, *Ralstonia*, *Cupriavidus*, *Achromobacter*, *Nocardia,* and *Alcaligenes*, as Gram-negative strains, and *Corynebacterium*, *Rhodococcus,* and *Bacillus* as Gram-positive strains, were used to confirm complete mineralization of biphenyl and some PCBs (Žerađanin et al., 2022). These microorganisms use Fe (III) nitrate, CO_2, sulfate, or other electron acceptors (chlorate, Mn, etc.). The inclusion of bacteria to remove hydrocarbons, pesticides, dioxins, and furans has been documented previously for a variety of chemicals, including PAH, dichloro diphenyl trichloroethane (DDT), and PCB. PAH-degrading bacteria isolated from mangrove sediments were studied and it was discovered that *Alcaligenes* sp. SSK1B, *Microbacterium* sp. BPW, *Ralstonia* sp. BPH, *Novosphingobium* sp. PCY, and *Achromobacter* sp. SSK4, *PCY* has the most adaptability in degrading different PAHs (Rastimesina et al., 2022). The quantity of *Azotobacter* spp. and *Mucoromycetes* was low, and there were no representatives of the *Actinomycetes* group (Takagi et al., 2020).

Microbe-Mediated Phytoremediation Microbes, such as bacteria and fungi, have been shown in studies to play a helpful role in destruction, growth promotion, and stress relief. Furthermore, microorganisms assist plant species to survive in soils with higher levels of contaminants. Microbes trigger intrinsic plant growth-promoting features such as phytohormone synthesis, siderophores, and chelating compounds, and hence indirectly contribute to bioremediation and microbe-mediated pollution clearance. These microorganisms mobilize contaminants in the rhizosphere of plants, which the plants then absorb. They also aid in the resistance of plants to various environmental conditions. The bioremediation activity is aided by the production of exudates and enzymes that drive biochemical and microbiological activity in the surrounding soil. The future challenge for the growth of microbe-based phytoremediation will be to create reliable tools for predicting their success in restoring contaminated sites (Rabani et al., 2020; Dhir et al., 2009; Malik et al., 2022).

The majority of microbial transformations of organic compounds are driven by a need to reduce toxicity, for which microbes may have to endure an energy deficit. As a result, the considerable energy given by root exudates may aid and drive the activities. Root exudates stimulate soil microbial populations, which benefits plants by increasing the availability of soil-bound nutrients and degrading phytotoxic soil pollutants. Soil microbes are also known to produce biosurfactant chemicals, which may help with the removal/breakdown of organic pollutants by increasing their availability to plants, as well as reducing their toxicity (Chaudhry et al., 2005). Arbuscular mycorrhizal fungus-assisted phytoremediation is an appealing approach to plant-based environmental remediation among polluted soil remediation technologies. Because of the benefits to their symbiotic partners, arbuscular mycorrhizal connections are vital in the restoration of damaged ecosystems. Arbuscular mycorrhizal fungi not only help plants to establish and survive in POP-contaminated soil by protecting them from POP phytotoxicity, but they also help with soil bioremediation by increasing telluric microbial activity and improving soil structure. (Lenoir et al., 2016). Plants paired with POPs-degrading rhizosphere and/or endophytic bacteria provide a viable technique for POPs-contaminated site cleanup. In bacteria-assisted phytoremediation of the POPs endophytic bacteria and rhizobacteria with relevant genes for pollutant breakdown, transformation, and mineralization help to reduce plant toxicity or phytovolatilization. Furthermore, using metagenomic approaches to learn about the metabolic activity and variety of bacteria can help us to create more sustainable bacteria-assisted phytoremediation solutions (Arslan et al., 2015).

11.8 Rhizoremediation

Rhizoremediation procedures are based on plant–bacteria interactions and can be utilized to effectively remove a wide range of contaminants from the environment while overcoming the limitations of individual phytoremediation. Akkaya et al. (2020) reported, 1 mM and 1.5 mM concentrations of 2,4-dinitrotoluene (2,4-DNT) degrading *Pseudomonas putida* strain KT.DNT and various growth stages of *Nicotiana tabacum* were tested in an in vitro tissue culture system to determine the best conditions for plant–rhizobacterium association for 2,4-DNT remediation. 2,4-DNT is a common soil contaminant in ammunition manufacturing facilities, originating mostly from the synthesis of the explosive 2,4,6-trinitrotoluene, but also from the manufacture of polyurethane foams and dyes. In comparison with non-inoculated plants, 5-day-old *N. tabacum* plants inoculated with 2106 cfu/mL bacterial inoculum for 3 weeks were favored for rhizoremediation tests because they demonstrated an almost threefold increase in fresh and dry biomass. When these seedlings were planted alone or with *P. putida* KT2440 or *P. putida* KT.DNT in soils polluted with 1 mM and 1.5 mM of 2,4-DNT, KT.DNT-inoculated tobacco plants showed maximal degradation rates of 98% and 93% after 14 days. Our findings suggest that using 2,4-DNT-degrading bacteria injected with *N. tabacum* plants to speed up and improve the remediation of 2,4-DNT-contaminated soil might be beneficial (Akkaya et al., 2020). Although bacteria catabolize a significant amount of organic contaminants in the absence of plants, this process is frequently inefficient owing to the relatively small microbial populations in bulk soil. Because plant–rhizobacteria interactions boost the proliferation and degradation capacity of applied bacteria through the rhizosphere environment of growing plants, the combined usage of plants and related bacteria strengthens the role of each partner. Inoculation of rhizobacteria, on the other hand, improves plant tolerance to pollution stress and speeds up response and biomass synthesis. Increased plant biomass production owing to bacterial inoculation would increase plant tolerance to phytotoxic pollutants, resulting in reduced plant stress and improved soil health. *P. putida* KT2440 has already been described as having the ability to tolerate a variety of aromatic chemicals. Furthermore, owing to the significant degradation capability of KT.DNT, plants inoculated with *P. putida* KT.DNT had a better tolerance to 2,4-DNT-contaminated soils (1 mM and 1.5 mM) than non-inoculated or KT2440-inoculated plants. It has been demonstrated that a plant-associated 2,4-DNT-degrading bacterium has a high potential for rapid 2,4-DNT breakdown in polluted soil (Akkaya et al., 2020; Tomer et al., 2021; Komives et al., 2009). The following plants were studied for POP uptake and translocation in both field and greenhouse conditions: spinach (*Spinacia oleracea*), peppers (*Capsicum annuum*), lettuce (*Lactuca sativa*), tomatoes (*Lycopersicon esculentum*), potatoes (*Solanum tuberosum*), carrots (*Daucus carota*), beans (*Phaseolus vulgaris*), corn (*Zea mays*), dandelion (*Taraxacum officinale*), beets (*Beta vulgaris*), eggplant (*Solanum melongena*), vetch (*Vicia villosa*), ryegrass (*Lolium perenne*), mustard (*Brassica juncea*), lupins, etc. (Ankit et al., 2020) (Fig. 11.2).

11.9 Case Study: Reduction of Phenanthrene in *Trifolium repens* L. by *Diaphorobacter* sp. Phe15

Although some root-associated bacteria have been shown to digest PAHs in polluted soil, their performance and dynamic distribution on the root surface and in plant tissues are as yet unknown. Polycyclic aromatic hydrocarbons are a type of harmful organic pollutant that can be found in large quantities in soil. The incomplete combustion of coal, petroleum, wood, and other organic matter produces PAHs. In this investigation, greenhouse container tests were carried out by inoculating the white clover (*Trifolium repens* L.) with the phenanthrene-degrading bacterium *Diaphorobacter* sp. Phe15, which was isolated from root surfaces of healthy plants polluted with PAHs. Phe15 colonization, dispersion, and performance in white clover were studied dynamically. Nutrients including potassium, nitrogen, and phosphorus are frequently given to the soil environment in actual soil bioremediation to encourage bacterial growth and PAH breakdown. Growing plants, on the other hand, provide a stable habitat and root exudates as nutrient substances for root-associated PAH-degrading bacteria, which can degrade PAHs more efficiently and

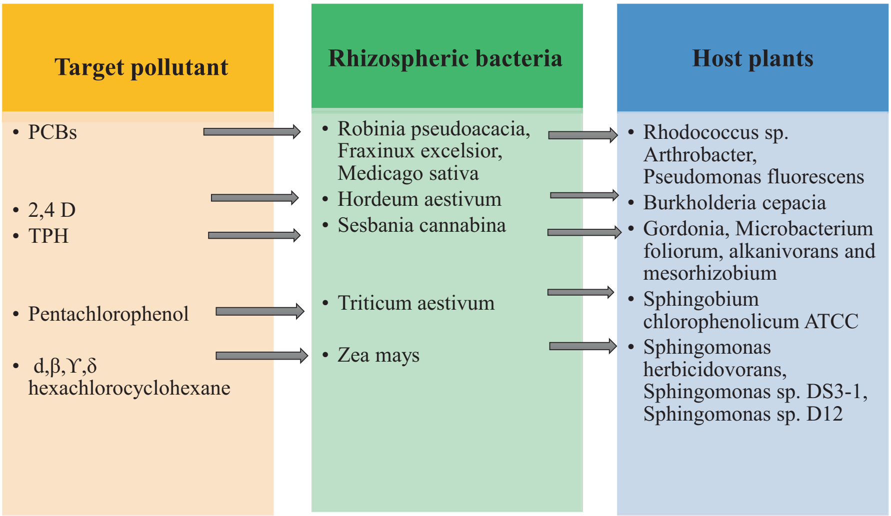

Fig. 11.2 POP degradation by means of a plant–rhizobacteria partnership

persistently, and plants can also metabolize PAHs, resulting in a higher removal efficiency than single inoculation of degrading bacteria. The effects of Phe15 colonization on phenanthrene metabolic enzyme activities and the molecular pathways involved in phenanthrene breakdown in inner plants, on the other hand, are as yet unknown (Zhao et al., 2022).

11.10 Fungal Degradation

Fungi are common organisms that play an important role in all ecosystems. These are microscopic eukaryotic organisms capable of growing on a variety of substrates and performing their function for a virtually endless amount of time. These are a varied group of organisms that are capable of removing waste from a variety of industrial wastewaters including yeast, filamentous fungi, and molds. The white-rot fungal cultures were discovered to be capable of not only degrading but also removing certain PAHs. Fungi also help in the decomposition of dyes, pesticides, PCBs, hydrocarbons, and chlorinated and phenolic chemicals by producing enzymes such as laccases, manganese peroxidase, and lignin peroxidases (Rastimesina et al., 2022; Gaur et al., 2018; Matsumoto et al., 2009).

Mycoremediation of PAH-contaminated oil-based drill cuttings using white rot fungi has recently been studied. The findings revealed that the fungi can biotreat PAH-contaminated oil-based drill cuttings. Many microalgae and protozoa are found in ecosystems and are involved in a variety of activities, but just a handful have been linked to the breakdown of hydrocarbons, pesticides, and PCBs. In their metabolic route, several microbes use aromatic hydrocarbon and nonchlorinated aliphatic as a carbon source. These hydrocarbons travel very slowly to the active site of the bacteria owing to their hydrophobic character. Algae that can degrade petroleum hydrocarbons into a less hazardous state do so with the help of enzymes that they manufacture. *Scenedesmus obliquus* GH2 was employed to build an artificial microalgae–bacterial consortium for crude oil degradation, and it was discovered that by the artificial consortium both aromatic and aliphatic hydrocarbons were destroyed efficiently (Gaur et al., 2018; Takagi et al., 2020).

11.11 Phytoremediation

Phytoremediation is a collection of unique technologies that use the abilities of plants to clean up toxins in the environment. Phytoremediation is a hot topic in science right now because of its enormous promise as a long-term alternative to existing techniques of recovering contaminated lands. It is widely utilized in mechanical and chemical cleanup approaches as it is a cost-effective and environmentally friendly alternative (Asante-badu et al., 2020; Echereme et al., 2018). Phytoremediation technology is the most successful remedial method for reducing POPs, and it has recently attracted a lot of interest because of its eco-friendliness, its widespread applicability, and its cost-effectiveness. However, only a few plants can absorb POPs from the soil, and plants lack fixed enzymes that can breakdown or detoxify POPs. Only a few plant species, such as *Cucurbita* species, have shown a specific ability to ingest significant amounts of POPs from the soil and are considered POP hyperaccumulators. Furthermore, genes that encode and express POP-degrading enzymes have been discovered and extracted from POP-degrading bacteria. The phytoremediation of POPs can be carried out by producing transgenic hyperaccumulator plants that encode POP-degrading enzymes. However, creating transgenic plants is a time-consuming and costly process (Malik et al., 2022). It was investigated whether tomato, alfalfa, soybean, and sunflower species could be used to remove endosulfan from the soil. The findings showed that, with the exception of soybean, the phytoextraction rate of all other species increased at 60 days as soil pesticide levels decreased. Sunflower plants showed the highest rate of phytoextraction (2.23%), followed by tomato (1.18%), soybean (0.43%), and alfalfa (0.11%). The spread of DDT residues in agricultural soils in southern Ontario, Canada, was investigated further (Asante-badu et al., 2020) (Table 11.1).

POP Uptake and Translocation in Plants The pollutant's concentration in the soil solution, its capacity to enter the root system, and the rate of transpiration in the plant all influence its uptake and translocation in plant tissues. The bioavailability of the pollutant limits the efficacy of the phytoremediation of POPs. The physical and chemical properties of POP molecules, as well as those of the soil, determine bioavailability. Water transport into the cells regulates the uptake of aqueous POP solutions and their transfer within plant tissues, which are normally passive processes. POPs can only be removed from soil by increasing their apparent water solubility (Ekperusi et al., 2018; Rabani et al., 2022).

Table 11.1 Phytoremediation of pesticides by various plant species previously documented

Sr no.	Plant species Scientific name	 Common name	Persistent organic pollutants	Plant uptake and target organ	Usage	Toxicity mg/kg	Persistence	References
1.	*Ricinus communis*	Castor bean/castor oil plant	Aldrin	Absorbed, roots	Insecticide	39	5 years	Asante-badu et al. (2020), Arslan et al. (2015)
			Endrin	Translocated, shoots	Pesticide	43.4	12–15 years	
			DDT	Absorbed and translocated, shoot	Pesticide	113–800	2–15 years	
			Hexachlorocyclohexane	Translocated Needles	Insecticide	88–190	Up to 2 weeks	
			Heptachlor, methoxychlor	Translocated, Shoot	Pesticide	40–162	Up to 2 years	
			Chlordane, chlorpyrifos	Absorbed, root	Insecticide	126–132	Up to 50 years	
2.	*Zea mays*	Maize/corn	Endosulfan	Translocated, roots	Pesticide	18–160	Up to 50 days	Rabani et al. (2022)
			Hexachlorocyclohexane	Translocated needles W	Insecticide	88–190	Up to 2 weeks	
			DDT	Absorbed and translocated, shoot	Pesticide	113–800	2–15 years	

(continued)

Table 11.1 (continued)

Sr no.	Plant species		Persistent organic pollutants	Plant uptake and target organ	Usage	Toxicity mg/kg	Persistence	References
	Scientific name	Common name						
3.	*Orychophragmus violaceus* L. O. E. Schulz	Chinese violet cress	DDE	Translocated, shoot	Pesticide	79.6	4.7– 6.1 years	Arslan et al. (2015)
			DDT	Absorbed and translocated, shoot	Pesticide	113–800	2–15 years	
			HCHs	Absorbed, root	Insecticide	88–190	Up to 2 weeks	
			DDT	Absorbed and translocated, shoot	Pesticide	113–800	2–15 years	
			DDE	Translocated shoot	Pesticide	79.6	4.7–6.1 years	
5.	*Achillea millefolium*	Common yarrow	DDT	Absorbed and translocated, shoot	Pesticide	113–800	2–15 years	Komives et al. (2009)
6.	*Acorus calamus*	Sweet flag – calamus	Atrazine	Absorbed, shoots	Herbicide	0.022	4 years	Asante-badu et al. (2020)
7.	*Amaranthus caudatus*	Love-lies-bleeding	Glyphosate	Translocated, roots	Herbicide	–	20 years	Asante-badu et al. (2020), Arslan et al. (2015)
8.	*Brassica campestris*	Mustard/turnip rape/keblock	Endosulfan	Translocated, roots	Pesticide	18–160	Up to 50 days	Asante-badu et al. (2020), Arslan et al. (2015)
9.	*Plantago major*	Broadleaf plantain/ white man’s foot/ greater plantain	Endosulfan	Translocated, roots	Pesticide	18–160	Up to 50 days	Asante-badu et al. (2020), Arslan et al. (2015)

In plants, POPs can enter through leaves and roots, but it is the roots that take them up and transport them to the aboveground sections. Because POPs are man-made compounds, they lack particular transporter proteins in plants; therefore, they are taken up by roots via simple diffusion through the cell wall, where they reach the xylem stream. Different plant types can absorb various POPs to varying degrees. To prevent the toxicity associated with ingested pollutants after uptake and translocation, plants normally use one of two procedures: evapotranspiration and/or phytodegradation. Evapotranspiration is the primary mechanism by which plants release pollutants into the atmosphere through their leaves for the majority of contaminants (Žerađanin et al., 2022; Lenoir et al., 2016).

Metabolism of POPs in Plants The conversion of pollutants by plant tissues into harmless derivatives would be an effective phytoremediation of POPs. However, despite the ability of plants to change a variety of xenobiotics via a variety of chemical/biochemical metabolic pathways, little is known about the rates of transformation and the structure of the metabolites generated from individual POP molecules by plant cells. Phases I and II are the two types of biotransformation reactions, with phase I involving oxidative transformations and phase II involving the conjugation of phase I products to sugars or amino acids. In plants, cytochrome P-450-containing mixed-function oxygenases usually mediate oxidative metabolism in the phase I system. Both xenobiotic substrates and Both xenobiotic and endogenous substrates are supported by the oxidative, peroxidative, and reductive metabolism of these enzymes. Hydrophobic xenobiotics functionalized by the phase I system are transformed to more hydrophilic forms in the phase II systems by conjugating them with sugars or sulfhydryl (−SH) group-containing tripeptides such as glutathione (-L-glutamyl-L-cysteinyl-glycine). Plants do not have the same excretion mechanism as animals. Toxic metabolites and contaminants are trapped in the cell vacuole in plant cells. This is an active phase III process that is catalyzed by membrane-bound ATP-driven pumps (Komives et al., 2009).

11.12 Techniques of Phytoremediation

When plants grow in soil contaminated with organic contaminants and heavy metals, they acquire or break down pollutants. An organic contaminant in plants is remediated by many natural biophysical and biochemical processes, similar to heavy metals. Depending on the compound, ambient circumstances, and plant genotypes, toxic substances can be stored, fixed, volatilized, transformed to various degrees, or a mixture of these processes (Asante-badu et al., 2020; Arslan et al., 2015).

Phytodegradation Phytodegradation (also known as pytotransformation) is the process of organic pollutants being taken in by plant cells and degraded or mineralized by enzymes. Nitroreductase (which decomposes nitroaromatic chemicals), dehalogenase (which decomposes chlorinated solvents and insecticides), and volatile enzymes are examples of these. Populus and sage are examples of plants possessing these enzyme systems. It has been claimed that *Armoracia rusticana*, for example, is able to breakdown benzophenone within its tissues. The green liver of the biosphere is made up of these green plants.

Phytostabilization An in situ technique called phytostabilization, also known as phytoimmobilization, in which harmful compounds are incorporated into the lignin of the root cell wall to generate nontoxic molecules, reducing the presence of pollutants in the agricultural and natural environment. The goal of this strategy is to keep pollutants from concentrating and dispersing in the soil. Many species of the genera *Alyssum*, *Haumaniastrum*, *Gladiolus*, *Eragrostis*, and *Asclepius* have been produced for this purpose. Because of their ability to release a greater number of chelating agents, these plant species can also help with phytostabilization. In other studies, dirty soil is covered with flora that is resistant to high applications of harmful components, minimizing soil erosion and pollutants leaking into groundwater. The contaminants are immobilized by these compounds, preventing them from being absorbed and decreasing their mobility in the soil. As a result, plants with stabilizing potential play an important role in agricultural productivity and vegetation restoration in contaminated areas. These compounds were known to restrict absorption, immobilize pollutants, and hence reduce their movement in the soil, according to reports.

Phytovolatilization Phytovolatilization refers to the ability of some green plants to absorb specific metals/metalloids and then vaporize Se, Hg, and other ions, take them up by the roots, convert them into nontoxic forms, and then release them into the atmosphere. The downside of phytovolatilization, on the other hand, is that harmful molecules released into the atmosphere can precipitate and re-deposit in the environment, resulting in the formation of new toxic substances. Transgenic plants (containing bacterial genes) such as *Brassica napus*, *Liriodendron tulipifera*, *Nicotiana tabacum* o *Arabidopsis thaliana* are among the plants being studied.

Phytoextraction This method describes how contaminants are absorbed by roots, then displaced and accumulated in air particles. By absorbing water available to plants, phytoaccumulation, also known as phytoextraction or hyperaccumulation, removes salts, metals, and organic compounds from the soil through cationic pumps and sorption. *Alyssum bertolonii*, *Elsholtzia splendens*, *Pteris vittata*, and *Thlaspi caerulescens* have all been identified as hyperaccumulating plants of Ni, Zn/Cd, As, and Cu according to research (Zhao et al., 2022).

Phytofiltration Through the root structure or additional submerged organs, plants absorb distillate and/or precipitate pollutants from the environment, particularly radioactive elements and heavy metals. Plants are stored in a hydroponic system through which wastewater runs and is "filtered" by the roots (rhizofiltration) or other structures that can absorb and concentrate pollutants (hyperaccumulators) or tolerate contaminants for the greatest results throughout this process. In hydroponics, *Elsholtzia argyi* and *Elsholtzia splendens* were found to be effective at removing Cu from contaminated water. Their findings suggest that *Elsholtzia argyi* has superior Cu phytofiltration (removal rate of 50–90%) to *Elsholtzia splendens* (removal rate of 45–80%), which is connected to increased Cu concentrations and shoot translocation (Akkaya et al., 2020).

Rhizodegradation (Phytostimulation) The breakdown of organic molecules in the soil via root microbial activity is known as rhizodegradation or phytostimulation. Improved biodegradation of pollutants by root-associated fungus and bacteria of a certain plant species. There are symbiotic mycorrhizal fungi that live on plant roots and are far more useful for nutrient production and biochemical availability. Biodegradable enzymes can also be released by the plants themselves. Because of the shifting spatial distribution of nutrients, the microbial interaction in the rhizosphere is varied; however, *Pseudomonas* species are the organisms most closely associated with the root (Takagi et al., 2020).

Phytodesalination According to the literature, this is a relatively new method of removing saline from internal salts. *Suaeda maritima* and *Sesuvium portulacastrum* have been shown to be very successful in eliminating and accumulating sodium chloride (NaCl) from highly salty soil, according to studies. *Sesuvium verrucosum* Raf. and *Bacopa monnieri* (L.) Wettst were evaluated for their ability to increase their chemical characteristics in salty soil for 240 days in a field. It was also discovered that this group of plants had a phytodesalination capacity of 1.21 t Na/ha, which helped to prepare the soil for crop growth (Asante-badu et al., 2020).

11.13 Aquatic Macrophytes

Free-floating, submerged, and semi-aquatic emergent aquatic and wetland plant species have gained worldwide importance. Inorganic and organic pollutants, heavy metals, explosives, and radioactive wastes are just a few of the toxins that aquatic macrophytes can remove or degrade. Organic pollutants can be sequestered, removed, and transformed by aquatic plant species. The uptake and accumulation of organochlorine, organophosphorus compounds, and chlorobenzenes by aquatic plants has been extensively researched. The availability of the contamination in its protonated form determines the pollutant's passive absorption (Dhir et al., 2009). In plants, the protonated form of the contaminant is regarded as the species available for biotic partitioning, which is then linked to compartmentalization and enzymatic transformation in vacuoles. Plants use quick physical and chemical reactions including complexation and interaction with cuticular and membrane components to sequester organic molecules such as halogenated organic chemicals. The metabolic routes by which aquatic plant species convert organic pollutants have been identified. When aquatic plants are exposed to organic compounds, they undergo degradation or transformation, which can be oxidative or reductive, resulting in the creation of metabolites, which are then ingested by covalent binding to the plants.

Dehalogenation reactions have been described, especially for halogenated chemicals such as hexachloroethane, dichloro-diphenyl trichloroethane, and others in phytoreduction reactions. Phytoreduction products are either oxidized into polar chemicals or covalently bonded to plant tissues (assimilated), although reduction products are always more concentrated than oxidation products in any plant species. Enzyme-mediated reductive transformation mechanisms in plants have been reported. *Elodea* also has a dehalogenase activity that reductively transforms hexachloroethane (HCA) into perchloroethylene. DDT was reduced to equivalent dichlorodiphenyldichloroethane analogs, plant-bound fractions, and other unknown compounds in studies with *Elodea*. Rapid sequestration by partitioning to the lipophilic plant cuticles, phytoreduction to less halogenated metabolites, phyto-oxidation, and assimilation into plant tissues as nontoxic products, presumably formed by covalent binding with the plant tissues, are all mechanisms involved in the removal of halogenated organic compounds from water by aquatic plant species (Dhir et al., 2009; Russell, 2005; Ankit et al., 2020). Aquatic macrophytes have proven to be an environmentally benign and effective technique for removing organic pollutants from aquatic habitats. *Trapa natans*, *Eichhorn crassipes*, *Pistia stratiotes*, *Lemna minor*, and *Elodea canadensis* are examples of aquatic macrophytes that can be employed to clean up contaminated waste and wastewater systems. Furthermore, these plants aid in carbon sequestration, and their biomass can be used to make bioenergy (biofuel) simultaneously (Ankit et al., 2020). Microorganisms have been used to help with pollution repair and degradation. However, incorporating plants, invertebrates, and other types of living things broadens the range of applications for living systems. This could be a broader field for determining the best strategy for biological treatment. Each method has advantages and disadvantages; nevertheless, converting it into a benefit for the environment involves the use of biotechnological expertise by generating a new strain or changing existing strains for improved results (Table 11.2).

11.14 Advantages of Bioremediation

One of the primary advantages of bioremediation is that is a solar-driven process.

1. Phytoremediation utilizes the natural plant mechanism to take up and clean pollutants. So, additional labor, operational costs, equipment expenses are not required.
2. Plants accumulate POPs and convert them into a less hazardous form, which prevents air and water emissions and secondary waste production treatment and cost expenses.
3. Deep roots of plants prevent water runoff and soil erosion and make sites more attractive choices with potential increases in biodiversity.
4. Phytoremediation can be incorporated into other remediation techniques to boost the cleanup of contaminant.

Table 11.2 Aquatic macrophytes with the potential of remediating persistent organic pollutants

Plants species		Pollutants	References
Free-floating	*Spirodela oligorrhiza*	Organophosphorus and organochloride compounds (o, p-DDT, p, p-DDT), chlorobenzenes	Dhir et al. (2009)
	Pista stratiotes	Organochlorines (DDT, DDE, aldrin, endosulfan, pyrethroids, oxytetracycline	Ekperusi et al. (2018)
	Lemna gibba	Phenol, 2,4,5-trichlorophenol	Ankit et al. (2020)
	Lemna minor	2,4,5-trichlorophenol, halogenated phenols	Dhir et al. (2009)
	Eichhornia crassipes	Ethion, dicofol, cyhalothrin, pentachlorophenol	Dhir et al. (2009)
Submerged	*Elodea*	Hexachloroethane, DDT, carbon tetrachloride	Dhir et al. (2009)
	Elodea canadensis	Phenanthracene, organophosphorus and organochloride compounds, chlorobenzenes	Ankit et al. (2020)
	Myriophyllum aquaticum	Simazine, o, p-2 DDT, p,p-2 DDT, HCA, CT, perchlorate	Dhir et al. (2009)
	Potamogeton crispus	Phenol	Ankit et al. (2020)
	Ceratophyllum demersum	Organophosphorus and organochloride compounds, chlorobenzenes	Dhir et al. (2009)
Emergent	*Scirpus lacustris*	Phenanthracene	Ankit et al. (2020)
	Pontederia cordata	Oryzalin (herbicide)	Dhir et al. (2009)

11.15 Disadvantages of Bioremediation

Besides continuous improvement and application in phytoremediation, it does have some limitations.

1. The prime limitation is the partnership of plants and rhizobacteria. Rhizoremediation depends upon root interaction and contact time with pollutants. The deeper the roots, the more likely a deep cleanup of the contaminants.
2. Phytoremediation is a time-consuming technique as plants take a long time to grow up, which slows down the cleanup time.
3. The selected plant must need to be tolerant to the interested contaminant and phytotoxicity.
4. Other drawbacks include seasonal variations that affect plant growth, a large space is required for planting, and the expected transmission of pollutants to nearby sites (Russell, 2005).

11.16 Comparison of Conventional and Advanced Techniques

There are different conventional methods that are used for the removal of POPs including solvent extraction, incineration, gas phase chemical reduction, land filling, adsorption, membrane filtration, etc. These methods do not completely remove the pollutants, are also expensive, and require proper man power and infrastructure. Another method of removing toxic pollutants from the soil, water and sediments is biodegradation including phytoremediation and rhizoremediation. It is a most sustainable process as it does not disturb the environment in which microorganisms are involved that convert the complex organic pollutants into the simpler state. Also, it is a cost-effective technique. It is considered eco-friendly and economically efficient, and the best way to remove problematic pollutants from the environment.

11.17 Future Prospect

The POPs are of great importance for some industries so they are not totally avoidable. The only solution to this problem is to adapt techniques that can degrade the POPs effectively. There is a need to deal with the removal of hexachlorocyclohexane (HCH) as it is the most dangerous of the POPs. Different techniques can be used to remove HCH, such as the advanced oxidation process, adsorption and bioremediation. Although bioremediation is effective too, as it can be easily inserted into the wastewater system, it is not as efficient as AOP and its technical demand is higher than other methods. The oxidation process produces many secondary pollutants that then require further steps to remove these pollutants, making it a less efficient process compared with adsorption and catalytic degradation, no process is 100% efficient. There is a need to understand every factor along with economic efficiency and technical know-how of the process. Future studies need to eliminate the shortcomings of the processes.

The most effective solution for removing the pollutants from the environment requires the isolation, characterization, and purification of enzymes. As it is time consuming there is a need to use other approaches such as metagenomic and metaproteomic. Different pathways that microorganisms use to degrade the pollutants are still unknown and can be identified by the use of many biological approaches including proteomics, transcriptomics, phenomics, and metabolomics. Nanoparticles can also be utilized to remove these organic pollutants and it can be done using hybrid nanoparticles that can then render highly toxic pollutants less toxic.

11.18 Conclusion

Owing to a lack of studies, there is a gap in identifying the actual source of POPs other than agriculture. In developing countries, waste disposal in uncontrolled environments and its burning also release large amounts of POPs into the environment. Poorly treated wastewater from different industries (pharmaceutical, dyeing and textiles, thermal power plant, paper mill) that is released into the streams is also a source of POPs. The presence of different pharmaceutical, personal care products, and cosmetic compounds in water is also a greater concern globally. The data assessment of POPs to identify the ecological risk showed that POPs travel to high-latitude atmospheres so there will be higher rate of adsorption of these pollutants in areas where temperatures are low. The application of pesticides for agriculture purposes can cause airborne pollution as they volatilize in the air, and likewise the practice of cooking, cooling, and heating in a closed infrastructure can cause the dispersion of pesticides that are used indoors. These POPs can accumulate on the dust or aerosol particles and can be a major source of POPs in living organisms as they inhale or ingest these particles. It is also reported by different studies that these pollutants alter the temperature, weather patterns, and carbon cycle, and hence result in climate change. These alterations in the environment led to the increase in the wet deposition of POPs and an increase in the contamination of atmosphere and water bodies. Exposure of humans to the toxic, persistent compounds can be direct or they can enter into the human body through inhalation, ingestion, food, and skin. In individuals of all age groups POPs are present but children are at a greater risk owing to their physiology and behavior. Different disorders have been reported such as cryptorchidism, prostate cancer, and testicular cancer when male fetuses are exposed to these organic pollutants called. These pollutants also decrease the sperm count and reduce male reproductive capacity. In women negative effects occur such as breast cancer, cystic ovaries, and endometriosis. Biological treatment of POPs is the most attractive option of all. Plant-based solutions also enhance biological diversity and value to the ecosystem. The most effective solution to removing pollutants from the environment requires the isolation, characterization, and purification of enzymes. As it is time consuming there is a need to use other approaches such as metagenomic and metaproteomic.

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Chapter 12
Multiple Adaptation Strategies of Plants to Mitigate the Phytotoxic Effects of Diverse Pesticides and Herbicides

Samrat Banerjee, Mehali Mitra, Pinaki Roy, Sreyashi Nandi, and Sujit Roy

Abstract The increase in the world's population in the twentieth century resulted in the subsequent increase in the demand for food. To enhance the constant supply of food for this large population and sustainable crop production, different types of agrochemicals such as fertilizers, pesticides, fungicides, and herbicides were used by farmers for decades. Pesticides are mainly categorized as herbicides, fungicides, and insecticides based on the target they killed. Pesticides and herbicides are designed to kill and prevent pests and unwanted weeds respectively. As their mode of action is not species specific, they often harm other organisms including crops in the agricultural field when used in excess amounts. Over time, insects and weeds become adapted and develop resistance to such chemicals, which necessitates the excessive amount of usage and development of new chemical compounds to protect crops. In many developing countries cheap compounds, such as dichloro-diphenyl-trichloroethane (DDT), hexachlorocyclohexane (HCH), and lindane are popular among farmers, even though they are environmentally persistent and have a toxic effect on soil flora and fauna. Thus, the pesticide and herbicide compounds have emerged as a new global concern owing to their several phytotoxic effects. Moreover, the development of leaf and crop growth rate, and the nutritive composition of seeds, specifically the content of proteins, fall sharply following pesticide treatment. The herbicides and pesticides cause several cytotoxic and genotoxic effects which ultimately challenge the stability of the plant genome through the production of reactive oxygen compounds. To combat these stress conditions, plants have evolved several biochemical, physiological, transcriptional, and epigenetic strategies that together help to maintain the growth and development of plants. In this present book chapter, we summarize the harmful effects of pesticides and herbicides on crop plants and the different strategies evolved by plants to combat these emerging stress compounds to sustain growth and eventually survivability.

S. Banerjee · M. Mitra · P. Roy · S. Nandi · S. Roy (✉)
Department of Botany, UGC Centre for Advanced Studies, The University of Burdwan,
Golapbag Campus, Burdwan, West Bengal, India

T. Aftab (ed.), *Emerging Contaminants and Plants*, Emerging Contaminants
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12.1 Introduction

Most of the countries in the world depend on agriculture for their supply of food. The prime objective of all research is centered around the aim of increasing the production of food to meet the requirements of the increasing population. However, an increase in production is challenging owing to the limitations of agricultural land. During the last few decades, a broad range of pesticides have been used in the agricultural fields to enhance agricultural production and minimize the loss of yield caused by pests (insects, and other animals), microorganisms and prevent the growth of unwanted plants (weeds) (Gill & Garg, 2014; Zaka et al., 2019; Pimentel et al., 1992).

From ancient times pesticides have been used, but their use increased rapidly after the Second World War (Gavrilescu, 2005). The use of pesticides has become the central aspect of modern agricultural practice in both developed and developing countries because pesticide usage enhances agricultural production and decreases post-harvest loss (Rosell et al., 2008). Although agricultural yield has increased in the last decades with the increasing use of pesticides but the frequent and continuous use of persistent, nonbiodegradable, and nonspecific pesticides is responsible for surface water contamination, groundwater contamination, soil contamination, effect on soil fertility, contamination of air, soil, and nontarget vegetation (Aktar et al., 2009; Niti et al., 2013; Jayaraj et al., 2016; Morillo & Villaverde, 2017). Moreover, the presence of residues of these chemicals in food crops especially (vegetables and fruits) are contaminated and these are very dangerous for the consumers. It has many adverse effects on the environment and human health. Pesticides enter into the food chain and start to bioaccumulate at the higher tropic level. Exposure of humans to these pesticides has been linked to acute and chronic illness. (Mostafalou & Abdollahi, 2012). Governments imposed a global ban and take multiple measures to reduce the use of persistent organic pollutants. Eight of these POPs were insecticides (endrin, heptachlor, mirex, toxaphene, aldrin, chlordane, dieldrin, and dichloro-diphenyl-trichloroethane, DDT); one of them was a fungicide (hexachlorobenzene, HCB) whereas the rest were dioxins (some of their by-products in pesticide production), polychlorinated biphenyls (PCBs) and polychlorinated dibenzofurans (PCDFs). They are toxic and biomagnified in the food chain (Rosell et al., 2008; Ali et al., 2014).

Plants take up pesticides in several ways. The pesticide may absorb by roots or leaf surface and water-soluble pesticides may enter at the time of the transpiration pull. Volatile pesticides may enter through stomata at the time of transpiration (Sharma et al., 2019a). Pesticide-mediated toxicity is responsible for necrosis, chlorosis, stunting, burns, and twisting of leaves. Pesticides may decrease the photosynthetic efficiency of plants owing to the degradation of chlorophyll by the formation of reactive oxygen species (ROS) (Sharma et al., 2019a). The weeds and insects may become resistant to particular chemicals because of the excessive use of herbicides and pesticides (Heap, 2018). Researchers have been studying 60 years of herbicide resistance at the phenotypic, physiological, and genetic levels to diagnose the scale of the resistance problem and develop strategies to delay its spread. The researchers found striking discoveries about plant adaptation – ranging from identifying the genes and small mutations that are responsible for resistance (Powles & Yu, 2010; Délye et al., 2013). The potential impacts of herbicide resistance provide a challenging model system to discover herbicide-resistant weeds in three areas of interest to ecologists – the genetic basis of adaptation, evolutionary constraints, and experimental evolution (Bianchi et al. 2016). The host plant resistance (HPR) provides the most effective, eco-friendly method of pest control and is considered to be a key component of integrated pest management strategies. The potential impact of HPR is the genetic ability of the plant to improve its survival and for this reason, identifying and developing HPR is now a major thrust area of plant breeding and agriculture (Sharma & Ortiz, 2002). As chemical pesticides are nonspecific and cause environmental pollution, in recent times pesticides and herbicides have been considered to be new emerging threats to sustainable agriculture practices. In this chapter, we particularly summarize the negative impacts of these agrochemicals on plant growth and development and then we tried to give a brief idea about the plant adaptation strategies against these compounds.

12.2 History of Pesticide and Herbicide Usage

The first-generation pesticides for the control of pests contain toxic compounds such as arsenic and hydrogen cyanide but the second-generation pesticides include synthetic organic compounds. The green revolution began in Mexico in 1944, an agricultural movement to obtain a higher yield to meet the need for food for the increasing human population. As high-yielding varieties were not widely resistant to pests, the green revolution required much greater use of pesticides than the traditional system (Zacharia & Tano, 2011; Rashid et al., 2010). The use of nontargeted chemicals not only showed an adverse effect on crops but also severely affected the plant communities. For this reason, selective herbicide came into existence to kill only the weeds. 2,4-D was chemically synthesized in 1941 and thereafter several 2,4-D salt and esters were developed. This was the beginning of the chemical era of the development of herbicides (Vats, 2015).

Major attention was first paid to pesticide production around World War II; in this period, several chemical pesticides were synthesized such as aldrin, DDT, BHC, 2,4-D, chlordane, and endrin (Kaur et al., 2019). Paul Muller discovered the modern pesticide DDT in 1939. Later, Muller won the Nobel Prize in Medicine for this discovery. But the use of DDT would not last long. The book, "Silent Spring" by Rachel Carson, described the harmful effects of DDT in the year 1962. As a result, several states banned the use of DDT (Abubakar et al., 2020). The usage of pesticides throughout the world is 2 million tonnes and the global consumption of pesticides includes the use of herbicides (47.5%), insecticides (29.5%), fungicides (17.5%), and other pesticides (5.5%). China, the USA, Argentina, Thailand, Brazil, Italy, France, Canada, Japan, and India are the top ten pesticide-consuming countries in the world (De et al., 2014; Sharma et al., 2019a, b). Half of the global pesticide is used in Asia, in which India is one of the leading nations (Nayak & Hitesh, 2021).

12.3 Classification of Pesticides

Pesticides are natural or synthetic agents widely used in agriculture, forestry, aquaculture, and the food industry to kill or reduce weeds, fungi, and other organisms. The term pesticide is used in a broader sense. Pesticides include insecticides, herbicides, fungicides, rodenticides, and algicides. Pesticides may be classified based on toxicity, function, chemical compounds, and mode of action (Rashid et al., 2010; Akashe et al., 2018; Nayak & Hitesh, 2021). Pesticide toxicity is dependent on two factors: dose and time. Depending on the dose and time toxicity is classified into two types: acute and chronic. Acute toxicity is defined as the toxicity of pesticides for a short time exposure to plants and animals. On the other hand, chronic toxicity is a delayed effect after exposure to pesticides (Akashe et al., 2018). Herbicides have several types such as systemic, nonsystemic, selective, and nonselective. Systemic herbicides are also called translocated herbicides as they are absorbed from the site of absorption and are translocated in plants through the vascular system. Systemic herbicides take a certain time to kill weeds. Nonsystemic herbicides are also called contact herbicides as they only kill the portion with which it comes into contact. They are used repeatedly to kill or inhibit the regrowth of underground plant parts. Selective herbicides are specific, they suppress specific plants, and do not affect other plants, whereas nonselective herbicides only act on the plants that come into contact with it (Vats, 2015). In the agricultural field, these chemicals are used in several ways, such as spray, powder, and oil solutions (Rashid et al., 2010). Several herbicides act as activators or inhibitors of several enzymes, for example, 2,4-D reduces or inhibits the nitrogenase (organisms used to fix atmospheric nitrogen gas) and phosphatase (used in the hydrolysis of organic P to inorganic P) activity (Table 12.1).

Table 12.1 Herbicides and pesticides classified according to their chemical group and mode of action on plants

Chemical group	Mode of action on plants	Example
Amide	Inhibits photosynthesis at photosystem II site A. Inhibits the C4 photosynthetic enzymes such as PEPC, MDH, PPDK, and RuBisCO. Increases MDA content and CAT, POD, SOD decreases in roots. Increases proline in root stem and leaves	Propanil, acetochlor, alachlor, fomesafen, *Rac*-metolachlor, *S-metolachlor*
Aryloxyphenoxypropionate	Inhibits acetyl CoA carboxylase	Fenoxaprop, fluazifop, quizalofop, clodinafop, diclofop, haloxyfop, propaquizafop
Aryl triazinone	Inhibits protoporphyrinogen oxidase	Sulfentrazone, carfentrazone
Benzothiadiazole	Inhibits photosynthesis at photosystem II site B	Bentazon
Benzoic acid	Inhibits indoleacetic acid transport	Dicamba
Carboxylic acid	Inhibits indoleacetic acid transport	Clopyralid, fluroxypyr, picloram, triclopyr
Chloroacetamide	Inhibits the synthesis of very-long-chain fatty acids	Acetochlor, alachlor, metolachlor, s-metolachlor, dimethenamid
Cyclohexanedione	Inhibits acetyl CoA carboxylase, increases MDA, APOX, POD, and total phenolics content, but decreases CAT and SOD in leaves	Clethodim, sethoxydim, alloxydim, butroxydim, cycloxydim, tralkoxydim
Dinitroaniline	Inhibits microtubule assembly. Decreases H_2O_2 MDA content in leaves and increases in total phenolics. At higher concentrations decreases germination percentage of some seeds	Benefin, ethalfluralin, oryzalin, pendimethalin, prodiamine, trifluralin
Diphenyl ether	Inhibits protoporphyrinogen oxidase	Acifluorfen, fomesafen, fluoroglycofen, lactofen
Imidazolinone	Inhibits acetolactate synthase. Increased MDA, APOX, CAT, GR, and GST content in leaves. Inhibit the C4 photosynthetic enzymes like PEPC, MDH, PPDK, and RuBisCO	Imazamethabenz, imazamox, imazapic, imazapyr, imazaquin, imazethapyr
Isoxazole	Bleaching: inhibition of 4-HPPD	Isoxaflutole
Isoxazolidinone	Inhibition of DOXP synthase and formation of pigment necessary for photosynthesis	Clomazone

(continued)

Table 12.1 (continued)

Chemical group	Mode of action on plants	Example
Neonicotinoid	Increase O_2^-, H_2O_2, MDA, and proline content in the seedling. Ascorbate, GSH, tocopherol, and total phenolics increase in leaves	Imidacloprid
Nitrile	Inhibit photosynthesis at photosystem II site B	Bromoxynil, ioxynil
N-Phenylphthalimide	Inhibits protoporphyrinogen oxidase and a negative impact on photosynthesis by the reduction in foliar chlorophyll, carotenoid content, and alteration of plastid structure	Flumiclorac, flumioxazin
Oxyacetamide	Inhibits the synthesis of very-long-chain fatty acids	Flufenacet
Organophosphorus	Increase in O_2^-, H_2O_2, MDA, APOX, GPOX, GR, GST, and POD in seedlings and shows significant reduction and delay in seed germination at higher concentrations. The foliar application increases plant height, the number of branches, and total leaf area	Chlorpyrifos
Organophosphorus	Increase in O_2^-, H_2O_2, MDA, OH, GSH, ascorbate in leaves	Dimethoate
Organophosphorus	Systemic pesticide, regulates the synthesis of amino acids. Inhibits 5-enolpyruvyl-Shikimate-3 phosphate synthase. Increase in MDA, proline, and GSH content in leaves and roots	Glyphosate
Phenoxy	Systemic pesticide acts as a growth regulator, disrupts protein synthesis, and hormonal balance. Inhibit indole acetic acid transport	2,4-D, MCPA, MCPB, mecoprop
Phenyl-carbamate	Inhibits photosynthesis at photosystem II site A	Desmedipham, phenmedipham
Pyrazolone	Bleaching: inhibition of 4-HPPD	Topramezone
Pyrethroid	MDA, proline, ascorbate, GSH content increased in leaves	Deltamethrin
Pyridazinone	Inhibit photosynthesis at photosystem II site A and inhibitor of the phytoene desaturase	Pyramin, norflurazon
Pyridine	Inhibits microtubule assembly and increase O_2^-, H_2O_2, MDA, proline content in leaves	Dithiopyr, thiazopyr, fluroxypyr

(continued)

Table 12.1 (continued)

Chemical group	Mode of action on plants	Example
Pyridinecarboxamide	Inhibitors of the phytoene desaturase	Diflufenican, picolinafen
Pyrimidinyl-thio benzoate	Inhibits acetolactate synthase	Bispyribac-sodium, pyrithiobac, pyribenzoxim
Quaternaryammonium	Contact pesticide, disrupts the cell membrane, Photosystem I – electron diversion. Decrease in chlorophyll content and increase in SOD, APX, and lipid peroxidation	Paraquat
Semicarbazone	Inhibits indoleacetic acid transport	Diflufenzopyr
Sulfonyl-aminocarbonyl-triazolinone	Inhibits acetolactate synthase	Flucarbazone-sodium, propoxycarbazone
Sulfonylurea	Inhibits acetolactate synthase	Amidosulfuron, azimsulfuron, bensulfuron, chlorimuron, chlorsulfuron, cinosulfuron, cyclosulfamuron, ethametsulfuron, ethoxysulfuron, flazasulfuron, foramsulfuron, halosulfuron, iodosulfuron, mesosulfuron, metsulfuron, nicosulfuron, primisulfuron, prosulfuron, rimsulfuron, sulfometuron, sulfosulfuron, thifensulfuron, triasulfuron, tribenuron, triflusulfuron
Thiocarbamate	Inhibits lipid biosynthesis – not acetyl CoA carboxylase inhibition	Butylate, cycloate, EPTC, esprocarb, molinate, pebulate, prosulfocarb, thiobencarb, triallate, vernolate
Triazine	Disrupts the photosynthetic electron transport by inhibiting photosystem II site A. Inhibits the C4 photosynthetic enzymes such as PEPC, MDH, PPDK, and RuBisCO. Increases O_2^-, H_2O_2, TBARS in leaves and GSH, ascorbate in the shoot	Ametryn, atrazine, cyanyzine, desmetryn, prometon, prometryn, propazine, simazine, simetryu, terbumeton, terbuthylazine, trietazine
Triazinone	Inhibits photosynthesis at photosystem II site A. Affects activity of nitrate reductase, nitrite reductase, glutamine synthase, and glutamate synthase	Hexazinone, metamitron, metribuzin
Triazolinone	Inhibits photosynthesis at photosystem II site A	Amicarbazone
Triazole	Inhibitors of carotenoid biosynthesis	Amitrole, aclonifen

(continued)

Table 12.1 (continued)

Chemical group	Mode of action on plants	Example
Triazolopyrimidine	Inhibition of acetolactate synthase	Cloransulam-methyl diclosulam, florasulam, flumetsulam
Triketone	Bleaching: inhibition of 4-HPPD	Mesotrione, sulcotrione
Uracil	Inhibits photosynthesis at photosystem II site A	Bromacil, terbacil
Urea	Inhibits photosynthesis at photosystem II site A. decrease in chlorophyll content. Affects the development of roots and leaves. Increases O_2^-, H_2O_2, MDA, TBARS, and proline content in leaves	Chlorotoluron, dimefuron, diuron, fluometuron, isoproturon, linuron, methibenzuron, metoxuron, monolinuron, siduron, tebuthiuron
Unclassified	H_2O_2 increases in leaves and roots and POD decreases in roots and leaves	Dichlorobenzene
Unclassified	Leaves and root H_2O_2 increase Root and leaves POD increase	Monochlorobenzene
Unclassified	Leaves and root H_2O_2 increase Root and leaves POD no change	Trichlorobenzene
None accepted	Inhibit glutamine synthetase	Glufosinate

12.4 Pesticide- and Herbicide-Mediated Phytotoxicity

12.4.1 *Impacts of Pesticide and Herbicide Compounds on Plant Growth and Development*

The multiple cropping associated with modern agricultural practices unavoidably intensifies the occurrence of disease incidence. Many pesticide compounds are applied in the agricultural field for the protection of crops from diseases and the reduction of unwanted plants. The presence of certain functional groups, such as −OH, −NH2, −NHR, −CO.NH2, −COOR, and −NR3, in the molecular composition of the pesticides, hastened adsorption, particularly in soil, which subsequently may affect plant growth and development (Misra & Mani, 1994). The abnormal growth and development of plants may be due to the inability of plants to absorb micronutrients, especially Zn, Cu, and Mn, from the rhizospheric soil. Moreover, the presence of pesticide residues in soil results in the reduction of water potential in soil and depolarizes the plasma membrane of the root cells which may affect the nutrient absorption from soil (Siddiqui & Ahmed, 2006; Wright, 1994). Therefore, it can be hypothesized that extensive use of systemic pesticides in agricultural land might exhibit negative effects on the growth of nontargeted host plants. However, pesticides also exhibit significant effects on leaf growth. It was observed that at higher concentrations, pesticides significantly reduce the leaf area ratio, other leaf growth components, and maximum crop growth rate (Siddiqui & Ahmed, 2006). The nutritive composition of seeds was also affected by the pesticide treatment. Plant seeds produced with a higher concentration of pesticide showed lower lipid and protein content than seeds produced in control sites. However, in adverse conditions, pesticides rapidly reduce protein, lipid, and carbohydrate metabolism, which may cause the lowering of the net assimilation rate (Siddiqui & Ahmed, 2006). Glyphosate applied alone or in combination with other herbicides showed reduced seed germination, seed diameter, and seedling vigor (Subedi et al., 2017). Similarly, pyraflufen-ethyl applied with glyphosate or saflufenacil resulted in decreased dehulling efficiency. Moreover, Carpenter et al. (2020) observed the effect of glyphosate on flowering time. The inflorescence length of *Verbena* spp. showed greater sensitivity under dicamba and glyphosate treatment than under atrazine (Egan et al., 2014a, b). Significant delays in flowering and reductions in flower or fruit production have been found for a variety of crop species treated with different herbicides including glyphosate (Dupont et al., 2018). Reductions in flowering were also observed under multiple field studies with different herbicides including glufosinate ammonium (Boutin et al., 2014), chlorimuron ethyl (Boutin et al., 2014), dicamba (Bohnenblust et al., 2016), and 2,4-D (Hatterman-Valenti & Mayland, 2005). Meanwhile, exposure to dicamba has been shown to decrease total floral area along with reductions in key plant species such as *Monarda fistulosa* and *Taraxacum officinale* (Egan et al., 2014a). Excessive and repeated application of the chemical pesticides retards growth of vegetative and reproductive organs, and severely affects various morphological and physiological efficiencies of several important crops (Tort et al., 2005; Rio et al., 2012; Aksoy et al., 2013; Kilic et al., 2015; Shakir et al., 2016).

12.4.2 Physiological Impacts of Pesticides and Herbicides

Treatment with different pesticides has a significant effect on enzymatic and non-enzymatic antioxidant activities in plants. In *Vitis vinifera*, acetochlor treatment in leaves showed decreased activity of some enzymatic antioxidants such as APOX, CAT, POD, and SOD (Tan et al., 2012) whereas alachlor treatment shows increased POD and SOD activity in *Lactuca* (Štajner et al., 2003). In *Zea mays*, cyclohexene significantly decreases CAT and SOD in leaves (Radwan, 2012). In a recent study, it was observed that the application of emamectin benzoate, alphacypermethrin, and imidacloprid pesticides at recommended concentration or higher dosage to tomato seedlings showed disturbed cell viability, cell injury along with the modulation in SOD, CAT, POD activity and increment of GR antioxidant (Shakir et al., 2018). Moreover, atrazine significantly increases the H_2O_2 and MDA content in *Zea mays* and *Vicia faba* (Hassan & Alla, 2005). Cui et al. (2010) observed a similar kind of response following the treatment with amide pesticides. Treatment with organophosphorus compounds in the seedlings of rice and *Vigna radiata* showed a significant increase in O_2^-, H_2O_2, and MDA (Sharma et al., 2012, 2015). Plants treated with glyphosate show an increased amount of electrolyte leakage (Moldes et al., 2008). Moreover, pesticides also affect various non-enzymatic antioxidants in plants. In *Brassica juncea,* treatment with imidacloprid pesticide significantly enhances ascorbate, GSH, tocopherol, polyphenols, and total phenolics (Sharma et al., 2016a, b, c, d). Moreover, proline content is also increased following treatment with various pesticides (Parween et al., 2012; Badr et al., 2013; Sharma et al., 2015).

Common herbicides such as acifluorfen when applied to soybean, bean, pea, cotton, and spinach cause leaf injury (Saladin & Clément, 2005) along with chlorosis, necrosis, and wilting and shrinking of leaves. In another study, exogenous application of insecticides such as organophosphate, carbamate, diphenylethane to wild tamarind plants affects dry matter, leaf area and leaf number, and inhibits the photosynthetic machinery (Dalzell & Mullen, 2004; Saladin & Clément, 2005). The pesticides also interfere with the photosynthetic electron transport system via binding with (QA) D1 protein and inhibit the electron transport chain by acting as a nonreducible analog of plastoquinone and blocking PQH_2. Moreover, a well-known herbicide, paraquat, interferes with photosynthesis via the generation of the free radical O_2^-. Moreover, phenylpyrrole pesticides are responsible for a decrease in carotenoid, chlorophyll, sucrose, and hexose content along with overexpression of some secondary metabolites (Scarponi et al., 2002; Saladin et al., 2003).

The dynamics of plant–herbicide interactions and expression of phytohormone-related genes were found to be correlated in many previous studies (Doğramaci et al., 2015; Han et al., 2014; Köster et al., 2012). Treatment with phytohormones such as abscisic acid, auxins, brassinosteroids, cytokinins, gibberellins, ethylene, jasmonate, and salicylate thus directly or indirectly regulates the plant responses to different herbicides (Cui et al., 2010; Zhou et al., 2015). Several genes related to herbicide detoxification are directly regulated by phytohormones (Kerchev et al., 2015). Transcriptomic and metabolomic analyses have also revealed the response of herbicides to phytohormone gene expression. In plants, foliar application of glyphosate activates molecular processes associated with auxin, ethylene, and gibberellic acid biosynthesis and shikimate signaling pathways (Doğramacı et al., 2014).

12.4.3 Cytotoxic and Genotoxic Effects of Different Pesticides

Various studies have suggested the negative impact of some pesticides at the cellular/genetic level, including the formation of genotoxicity, DNA damage, and chromosomal abnormalities (Anitha & Savitha, 2013; Fatma et al., 2018). These pesticide compounds can increase the rate of mutation in plants although the detailed mechanism of action is still unclear. Recently, it was found that the application of mancozeb, imidacloprid, and sulfentrazone, on the model plant *Allium cepa*, showed cytotoxic and genotoxic effects by inducing different types of chromosomal abnormalities, likely sticky, disoriented and fragmented chromosomes, abnormal DNA condensation, and chromosome coiling by spindle inactivation and thus reduced mitotic index (Bianchi et al., 2016; Fatma et al., 2018). Similarly, a negative effect on the chromosome structure of *Vicia faba* was also observed under pesticide treatment (Devi et al., 1991). Recently, Mahapatra et al. (2019) demonstrated the cytotoxic and genotoxic effects of two pesticides, thiabendazole and tricyclazole, on *Trigonella*. They showed that treatment with pesticides not only produces chromosomal abnormalities but also alters the chromatin structure and induces DNA damage. Moreover, the components of pesticides negatively influence plant growth through genotoxic damage via the production of ROS molecules (Sies, 2015). The herbicides metribuzin, tribenuron-methyl, fenoxaprop-P-ethyl show a dose-dependent inhibitory effect on the mitotic index of the root meristem of *Triticum aestivum* (Menzyanova et al., 2020). The systemic herbicide Roundup, used to control weeds in cereal crops, increases the number of metaphase disorders in barley, which is due to the negative effect on the mitotic spindle formation (Truta et al., 2011). The negative impact of pesticides such as chlorpropham (Eleftheriou & Bekiari, 2000); imazethapyr (Rad et al., 2011); 2,4-D-isoproturon (Kumar, 2010) on the mitotic index of wheat has been already reported in various research works. Another study on the effect of the insecticide ethion on *Allium cepa* roots showed anti-mitotic activity in a dose-dependent manner along with the increase in various chromosomal aberrations such as nonsynchronized condensation of the chromosome, disturbed prophase, equatorial plate shifting, sticky chromosomes, C-metaphase, scattered prophase, and sticky metaphase (Lamsal et al., 2010).

12.4.4 Herbicides and Pesticides Affect Secondary Metabolite Biosynthesis

Secondary metabolite compounds in plants have many crucial functions related to plant growth and development, intra- and interspecific interactions, and plant defense against multiple stresses (Akula & Ravishankar, 2011; Jan et al., 2021). There are a few herbicides such as alachlor and glyphosate that affect specific biosynthetic steps. Alachlor specifically blocks the synthesis of flavonoids at later stages whereas glyphosate blocks the formation of all cinnamate derivatives by inhibiting EPSP synthase (Lydon & Duke, 1989; Velini et al., 2010). It was also observed that alachlor can reduce the lignin content of maize coleoptile (Hickey & Krueger, 1974). Even the micromolar level of alachlor can reduce anthocyanin and lignin biosynthesis. Moreover, herbicide compounds at sublethal concentrations may affect secondary metabolism either directly or indirectly. It was observed that dichlobenil, amitrole, acifluorfen, and paraquat can augment phenyl ammonia lyase (PAL) synthesis, resulting in an increase in phenol derivatives. The aromatic amino acid metabolism and phenylpropanoid biosynthesis was found to be altered in spinach leaves after treatment with acifluorfen (Komives & Casida, 1982). Meanwhile, in soybean seedlings a reduction in PAL biosynthesis was observed under treatment with triazine, urea, amide, and carbamate classes. The higher concentration of pesticides may induce the expression of phenolic compounds such as isoflavones, hydroxycinnamic acid, and polyphenolic compounds in soybean (Hoagland, 1989). In some cultivars of the bean, production of phytoalexin and cholesterol was significantly increased by atrazine, oxadiazon, and edothal. In barley seedlings, glyphosate inhibits flavonoid biosynthesis, even in the presence of the exogenous application of aromatic amino acids. Moreover, glyphosate is also responsible for the reduction in the medicarpin, glyceollin, and phaseolin in lucerne, soybean, and bean respectively, which is responsible for the reduction in fungal disease resistance. Another study revealed that some p-nitro substituted diphenyl ether (DPE) herbicides cause several-fold increases in a variety of secondary metabolites (Lydon & Duke, 1989). Moreover, herbicidal treatment with glyphosate may have an inhibitory effect on the formation of the alkaloids such as hyoscyamine and scopolamine (Zhang et al., 2021). However, concentrations of overall tropane alkaloid biosynthesis (tropinone and tropine) are significantly increased in *Datura* treated with these herbicides such as glyphosate (Deng, 2005).

12.4.5 Herbicides and Pesticides Severely Affect Ecosystem and Plant Communities

From various studies, it was observed that herbicides severely affect many plant communities and are responsible for the reduced flowering of many key species in the community (Egan et al., 2014a, b). The frequency of availability of many species significantly decreased following exposure to herbicides. Within the 3 years of constant application of atrazine and tribenuron-methyl species composition has been significantly altered and species diversity in the fallow field reduced. The different levels of sensitivity in plants to herbicides such as atrazine and tribenuron-methyl depend on the different chemical properties, weeding mechanisms, and weed control targets (Boutin et al., 2012; Tang et al., 2014). The diversity of the weed community is highly affected by different herbicides. Furthermore, the response of plant communities to different herbicides is often associated with community characteristics (Ortega & Pearson, 2011).

Pollinators are extremely important for global agricultural production. Many vegetable and fruit-producing crops require insect-mediated pollination, which can improve crop yield and quality (Garrat et al., 2014). Many crops, especially fruits and vegetables, require pollination, and emerging evidence suggests that insect-mediated pollination can improve yield and quality over self-pollination (Garrat et al., 2014). For instance, some herbicides may have direct or indirect effects on pollinators and they are sometimes lethal. For example, the honeybee population is greatly reduced following the application of paraquat (Moffett et al., 1972). Moreover, reduced numbers of flowers following treatment with different herbicides resulted in the reduction of pollinator visitation, which may further affect the pollinator community.

12.4.6 Plant Adaptations to the Adverse Effects of Pesticides and Herbicides

Pesticides and herbicides are applied all over the world to provide protection to plants from pests and inhibit the growth of unwanted plants. However, the growth and development of plants are sometimes negatively affected owing to the application of these pesticides and herbicides. Unfortunately, extensive use and reliance on pesticides and herbicides in the agricultural system cause environmental contamination. When this contamination surpasses the considerable threshold, biological communities are affected, which leads to environmental pollution (Moriarity, 1983). Pesticide toxicity reduces protein content and photosynthetic efficiency, which leads to lower chlorophyll content. ROS are also generated by the application of pesticides and herbicides, which results in oxidative stress to plants. On the other hand, herbicides, by their toxic mode of action, inhibit cell division, amino acid production, photosynthesis, and mimic natural plant hormones, which eventually cause deformities (Ross & Childs, 1995). The structural associations of these pesticides and herbicides have not yet been identified; thus, it is still challenging to identify the molecular site of action (Duke, 1990). However, to attenuate these negative effects of oxidative stress caused by the pesticides, the antioxidant defense system of plants is activated. Apart from this, epigenetic regulation may also contribute to herbicide resistance in plants. Epigenetic modification involves the process of modification of specific genetic elements, presenting an important role in reprogramming gene expression, not changing the DNA sequence. Epigenetic modifications can be a spontaneous mode of action or they can be induced genetically or environmentally. Sometimes the epigenetic alterations are reverted after any kind of stress exposure, but in some cases the changes are carried over for multiple generations causing trans-generational segregation, thereby possessing a selective benefit over those generations.

To avoid crop damage, pesticides are used to protect crops in the field as well as during post-harvest storage (Tomer, 2013). Pesticides are used to control a range of pests that attack crop plants, including soil insects, cutworms, leaf rollers, and aphids, among others (Goh et al., 2011). There are various options for controlling these insect pests, such as using bio-pesticides and developing pest-resistant transgenic cultivars. Chemical pesticides, on the other hand, are still the greatest and most extensively used approach for protecting crops from pests and resulting in good crop yields. Two million tons of pesticides are consumed each year around the world, according to reports (De et al., 2014). Herbicides, insecticides, fungicides, and other pesticides account for 47.55%, 29.5%, 17.5%, and 5.5% of global pesticide use respectively. Pesticide use has been demonstrated to have a negative impact on plant growth and development (Sharma et al., 2015, 2016a; Shahzad et al., 2018). The formation of ROS by pesticide treatment induces oxidative stress in plants (Sharma et al., 2018). This oxidative stress induces the breakdown of chlorophyll pigments and proteins, resulting in a decrease in plant photosynthetic efficiency (Xia et al., 2006; Sharma et al., 2015). The antioxidative defense mechanism of plants, which includes both enzymatic and non-enzymatic antioxidants, is activated in response to oxidative stress (Xia et al., 2009; Sharma et al., 2015, 2016b, c, d). Antioxidant defense system activation aids in ROS scavenging and lowers the oxidative stress induced by pesticide toxicity in plants (Sharma et al., 2015, 2017a, b). Plants have evolved the ability to reprogram gene expression in response to abiotic environmental stressors. Change in the chromatin state, as previously mentioned, is one of the processes that modulate transcriptional responses (Asensi-Fabado et al., 2017). Many herbicides cause oxidative stress in plants, which is similar to some abiotic stressors (Radwan, 2012). As a result, most of the enzymes engaged in herbicide metabolism are also implicated in plant stress response pathways, including the detoxification of ROS produced by stress (Iwakami et al., 2014; Cummins et al., 2013). However, it is unknown whether the herbicide's stress can create transient or even permanent epigenetic alterations. Because epigenetic modulation could contribute to herbicide stress transitory survival, this knowledge could change how resistance is classified based on an exclusively 'inherited' trait (Gressel, 2015). Changes in gene expression or the activity of transposable elements, which can potentially modify gene expression or even lead to gene duplication, could help explain the quick adaptability of weeds to pesticide selection pressure through epigenetic mechanisms. Most DNA methyltransferases, histone methyltransferases, and DNA demethylases were variably regulated in response to atrazine in rice, according to a recent study (Lu et al., 2016). The activation of certain genes involved in atrazine detoxification was aided by epigenetic changes. Different glyphosate dosages raised global DNA methylation levels in *Triticum aestivum* from 28 to 74% (Nardemir et al., 2015). Sublethal glyphosate dosages caused differently methylation areas across the *A. thaliana* genome in a dose-dependent manner. Surprisingly, additional abiotic stresses did not affect >90% of the impacted genes, implying that many epigenetic modifications are stress specific (Kim et al., 2017). These epigenetic pathways could be linked to the 'flipped on' or 'flipped off' control of genes involved in herbicide detoxification, such as P450s, GSTs, and ABC transporters, or transcription factors. The herbicide‘s signal (regulatory cascade) may also cause metabolite alterations, which result in changes in the expression and/or activity of RNA-directed DNA methylation, histone variations, and histone modification enzymes, all of which cause epigenetic changes. Changes in epigenetic enzymes and pathways can result in changes in genes implicated directly or indirectly in herbicide resistance, which can be passed down to future generations. When the plant overcomes the stress, some of the mutations return, indicating temporary changes associated with acclimation. Other changes, on the other hand, can be preserved mitotically and/or meiotically, resulting in a 'stress memory.' If mitotic heredity is present, the current generation will experience 'stress memory,' and if both mitotic and meiotic heritability are present, one or more generations may experience transgenerational persistence. Herbicide-resistant biotypes are selected in significant part because of the huge number of individuals subjected to severe selection pressure. Selecting for rare target-site resistance alleles (Gressel, 1995, 2011) largely associated with mutations in the gene that codes for the herbicide target enzyme are thought to result in high resistance levels when high herbicide doses are used. Sublethal herbicide doses, on the other hand, may select for resistance through the recurring enrichment of many minor additive genes (Gressel, 1995, 2011) associated with nontarget site resistance. Weeds in crop fields frequently receive sublethal herbicide doses as a result of drift, insufficient coverage, overtopping crop canopies, or purposely applied low rates. Low dosages can increase the frequency of plants with various alleles over time, resulting in a reduction in herbicide sensitivity at the population level. This happens more quickly in cross-pollinated species that can quickly accrue resistance genes (Powles & Yu, 2010; Délye, 2013). Herbicide-resistant biotypes emerge as a result of repeated cycles of selection by sublethal herbicide dosages. After some selection cycles, sublethal rates of ACCase herbicides boosted mean population survival over generations of *Lolium rigidum* (Heap, 1988; Neve & Powles, 2005). The use of a series of ACCase herbicides increased resistance in *L. multiflorum*, but it was not passed down to their offspring, indicating the nature of acclimation (Vila-Aiub & Ghersa, 2005). In *A. thaliana*, spontaneous mutation rates ranged from 7×10^{-9} to 2.2×10^{-8} base changes per site per generation (Ossowski et al., 2010; Willing et al., 2016). Under normal conditions, methylation cytosines are more likely to be mutated than nonmethylated cytosines (Willing et al., 2016). A total of 114,287 CG single methylation polymorphisms and 2485 CG differentially methylated areas that differed from the ancestral state were discovered in *A. thaliana* plants derived from single-seed descendants after 30 generations of study, contributing to phenotypic diversity (Schmitz et al., 2011). Although this is a large number, it is important to remember that methylation-based silence is rarely dependent on a single (non)methylated site (Pecinka et al., 2013). As a result, the majority of loci with polymorphic DNA methylation sites have no discernible effect. The frequency of DNA methylation loss or gain over a broader region with a potential phenotypic change appears to be similar to that of DNA mutations in *Arabidopsis* (Schmitz et al., 2011). Herbicide susceptibility was found to be elevated in various *A. thaliana* mutants linked to specific epigenetic DNA and histone methylation pathways, particularly for ROS1 (Repressor of Silencing 1). ROS1 is a DNA demethylase that removes DNA methylation for dynamic transcriptional control and can contribute to the increased expression of particular genes (Lei et al., 2015). This shows that epigenetic pathways may play a role in the control of herbicide detoxification genes. Furthermore, the intra-generational stability and transgenerational potential of these epigenetic modifications must be investigated to determine their ability to confer herbicide resistance throughout time. Epigenetic regulation of gene expression may play a role in the rapid local response to herbicide stress.

12.5 Conclusion

Indiscriminate use of agrochemicals such as pesticides and herbicides poses a serious threat to target as well as nontarget crop plants. The persistent nature of pesticides makes them highly toxic for both the plants and animals. The pesticides and

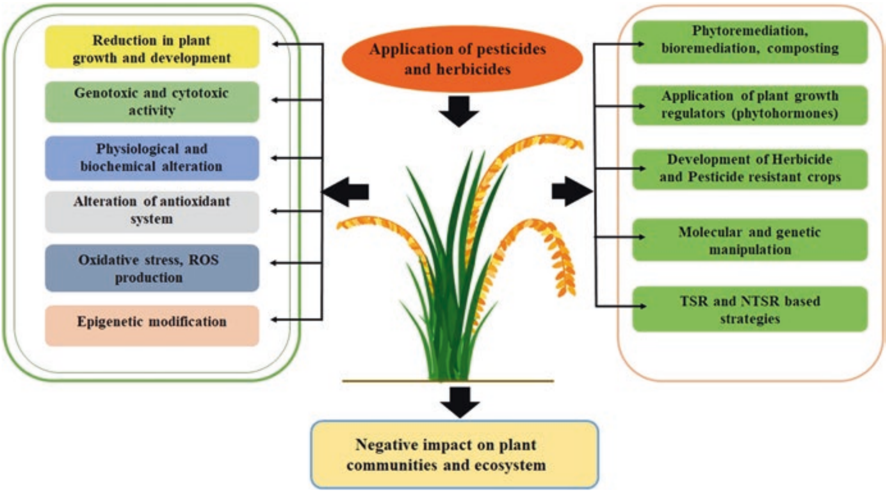

Fig. 12.1 Schematic representation showing the negative effect of pesticides and herbicides in crop plants. These chemical compounds severely affect plant communities and result in loss of biodiversity. The right-handed green boxes show the possible strategies for overcoming the toxicity caused by pesticides

herbicide compounds are bioaccumulated and biomagnified at successive trophic levels in an ecosystem, ensuring their long-term presence within it. After entering into the plant system, these agrochemicals severely affect growth and development and genomic stability by altering antioxidant activity and the generation of reactive oxygen compounds (Fig. 12.1). Furthermore, application of pesticides in an uncontrolled way may affect the biodiversity of a plant community. Plants develop multiple responses such as alteration of antioxidant enzyme activity, epigenetic modifications, and remodeling of chromatin to combat these emerging stresses. However, various strategies have been already applied such as phytoremediation, regulating the antioxidant system, breeding approaches for the enhancement of resistance, application of phytohormones, and organic farming to increase tolerance against these toxic compounds (Fig. 12.1). However, very little information is available regarding available molecular and genetic approaches. Extensive research is required for the identification of cellular defensins and kinases that are activated in response to pesticide toxicity. Moreover, safe methods such as the application of biopesticides in the agricultural field have to be taken into consideration. Extensive research work is still required to develop crops with increased tolerance to pesticides and herbicides, maintaining environmental safety and sustainable development.

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Chapter 13
Carbon-Based Hybrid Materials for Remediation Technology

Neetu Talreja, Divya Chauhan, and Mohammad Ashfaq

Abstract Emerging contaminations (ECs) have been discovered in water (municipal, drinking, and groundwater), food sources, soil, and aquatic bodies. These ECs may be from industrial, agriculture, hospital, and laboratory waste. ECs might alter reproductive functions in males and females, increase the chance of breast cancer, alter growth patterns, cause changes in immune function, and delay neurodevelopment in children. In this context, several nanomaterials (NMs), mainly carbon-based nanomaterials (CNMs) like carbon nanotubes (CNTs), carbon nanofibers (CNFs), and graphene have been effectively used in the removal of ECs from water due to their unique characteristics. This chapter focuses on synthesizing CNMs using the chemical vapor deposition (CVD) and liquid-phase exfoliation processes. We also discuss the removal ECs, mainly pharmaceutical compounds, endocrine disruptor compounds, and personal care products based on metal incorporation, surface functionalization, and polymeric composite materials.

Keywords Emerging contamination · Graphene · Personal care products · Endocrine disruptor compounds · Pharmaceutical

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N. Talreja
Faculty of Science and Technology, Department of Science, Alliance University,
Bengaluru, Karnataka, India

D. Chauhan
Department of Drinking Water and Sanitation, Ministry of Jal Shakti, New Delhi, India

M. Ashfaq (✉)
University Centre for Research & Development (UCRD), Chandigarh University,
Mohali, Punjab, India

Department of Biotechnology, Chandigarh University, Mohali, Punjab, India

T. Aftab (ed.), *Emerging Contaminants and Plants*, Emerging Contaminants
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13.1 Introduction

Presently, researchers are continuously focusing on the newly identified compounds or emerging contaminants (ECs) of anthropogenic origin. The ECs, also known as micropollutants generated through numerous sources, including synthetic and natural substances, has become a significant concern for the global population due to the likely threat to the environment and human health. ECs have been discovered in water (municipal, drinking, and groundwater), food sources, soil, and aquatic bodies (Rasheed et al., 2019; Patel et al., 2019b; Arruda et al., 2022; Ginebreda, 2021; Jari et al., 2022). These ECs may be from industrial, agriculture, hospital, and laboratory waste. The ECs might be categorized mainly into three types: (1) pharmaceuticals, (2) personal care products, and (3) endocrine disrupting compounds. According to the world health organization (WHO), ECs might alter reproductive functions in males and females, increase the chance of breast cancer, alter growth patterns, cause changes in immune function, and delay neurodevelopment in children (Diamanti-Kandarakis et al., 2009; Samal et al., 2022; Petrie et al., 2015; Imparato et al., 2022; Ortúzar et al., 2022). Therefore, there are urgent requirements for removing ECs from water bodies and soils.

Numerous processes such as adsorption, membrane separation technology, biological treatments, coagulation, precipitation, and advanced oxidation process (photocatalysis) have been used to treat environmental pollutants, including ECs (Ashfaq et al., 2022b, 2022c; Sultana et al., 2022; Afreen et al., 2022a; Omar et al., 2022). Among them, adsorption and advanced oxidation processes were considered appropriate candidates for the removal/degradation of ECs from water. With the help of nanomaterials (NMs), the efficiency of these processes might be significantly improved because these processes mainly depend on the materials. The changes in the characteristics of the material and the removal/degradation/sorption ability of the processes dramatically increased. Moreover, NMs show superior activity compared to bulky materials due to improved surface-to-volume ratio and reactive sites (Afreen et al., 2022b; Chauhan et al., 2022). Therefore, NMs might provide newer insight into the removal/degradation/sorption of ECs from water.

Several adsorbents (metal and metal oxides, activated carbon, carbon nanotubes (CNTs), carbon nanofibers (CNFs)) and photocatalytic materials (bismuth halides, graphene, graphene oxide, MXene, and WS_2) have been used for the removal/photodegradation of ECs from water (Ashfaq et al., 2022d; Talreja et al., 2021a; Ashfaq et al., 2021; Sasidharan et al., 2021; Talreja et al., 2021b, 2021c). Among them, carbon-based nanomaterials (CNMs) have the potential ability to easily remove/degrade ECs from water due to their unique characteristics like high surface area to volume ratio and high electrical and mechanical properties. The CNMs present an assortment of structures and extraordinary property mainly related to the unique atomic structure of the carbon atom. Moreover, based on shape, size, and dimensionality, the CBNs are arranged as zero-dimensional (0D), one-dimensional (1D), two-dimensional (2D), and three-dimensional (3D) (Ashfaq et al., 2022a; Patel et al., 2019a; Mahor et al., 2021; Maiti et al., 2019; Ma et al., 2021). The CNTs and

CNFs are one of the most important 1D nanomaterials that are extensively used in numerous applications like water treatments, agriculture, wound healing, drug delivery system, sensor, and energy (Kumar et al., 2011; Talreja et al., 2014, 2016, 2020; Saraswat et al., 2012; Kumar & Talreja, 2019; Song et al., 2018; Ashfaq et al., 2014, 2016, 2017a, 2017b, 2018; Afreen et al., 2018). Moreover, metal/polymers/surface functionalization incorporation within the CNMs or hybrid-CNMs (H-CNMs) might improve the applicability toward the end applications. The higher performance of H-CNMs is due to their improved characteristics like surface area and porosity. Additionally, incorporating metals, polymers, and surface functional groups has improved the selectivity and sorption ability. This chapter focuses on the synthesis of CNMs, adsorption of ECs, and photocatalytic degradation of ECs using CNMs-based hybrid materials. We also discuss the toxicity aspects of CNMs and future prospects in detail.

13.2 Synthesis of CNMs

Numerous processes have been developed to fabricate CNMs like CNTs, CNFs, graphene, etc. Researchers continuously focus to develop newer synthesis processes to fabricate low-cost CNMs. There are mainly two processes that have been commonly used for the synthesis of CNMs: (1) chemical vapor deposition (CVD) and (2) liquid-phase exfoliation process.

1. *CVD Process*

The CVD process is a scalable way to produce numerous materials like CNTs, CNFs, and graphene, mainly used in various applications, including electronics, sensors, and environmental remediation. Moreover, the reproducibility and variability in the different reactions using similar reaction conditions (temperature and carrier gas flow rate) is one of the most significant challenges. Additionally, it is difficult to optimize reaction conditions, mainly temperature, pressure, and carrier gas flow rate, for the synthesis of high surface area materials with desired density defects. The CVD reactor synthesizes numerous CNMs like CNTs, CNFs, and graphene (Georgakilas et al., 2016; Fauzi et al., 2018). The construction of CVD reactors is mainly quartz tubes that can sustain high temperatures.

2. *Liquid-Phase Exfoliation*

Usually, there are two types of liquid-phase exfoliation processes: (1) cavitation and (2) shear force in a mixer. The liquid-phase exfoliation is the straightforward process for synthesizing nanomaterials like graphene with an extensive production of nanomaterials. Moreover, the liquid-phase exfoliation process does not require high temperature/pressure/vacuum systems. However, large-scale application of the sonication process has been delayed due to low concentration and high energy consumption in production.

Normally, each process has established qualities and limitations that depend on its proposed applications. The encouragement of CNTs, CNFs, and graphene as next-generation materials for the fabrication/development of light-weighted, higher sensitive, small portable electronic devices, and portable devices for the detection of environmental pollution, especially ECs has been predicted for the future development of the society and build the nation.

13.3 CNMs for the Removal of ECs

Safe drinking water is essential to human life. Regrettably, contamination of water bodies has increased with increasing population and industrialization. Moreover, in addition to traditional contaminants, improving the ECs (pharmaceuticals, personal care products, and endocrine disruptors) in water bodies has become one of the greatest challenges. The ECs accumulate in the environment, affecting the environment and subsequently human and animal health (Sabzehmeidani et al., 2021). In this context, many CNMs like CNTs, CNFs, graphene, and graphene oxide have been used to remove/degrade ECs from water bodies.

13.3.1 Removal of Pharmaceutical Compounds

The ECs mainly use pharmaceutical compounds that accumulate in the water system that limit their water supply, such as drinking water. The accumulation of these compounds in water required effective treatments, as these compounds are highly toxic and affect human health. Numerous CNMs like CNTs, CNFs, graphene, graphene oxide, and fullerenes have been used to treat/remove pharmaceutical compounds from water. For example, Saraswat et al. synthesized nickel-doped carbon beads (Ni-CB) using the suspension polymerization method, followed by carbonization and activation using steam. The data suggested that the prepared Ni-CB effectively removes vitamin B_{12} from water using batch and continuous adsorption modes with ~300 mg/g sorption (Saraswat et al., 2012). Galhetas et al. synthesized fly ash from pine residue to remove acetaminophen. The data suggested that the prepared fly ash has a high surface area of ~1171 m^2/g. The high sorption ability of ~256 mg/g was observed mainly due to the high surface area (Galhetas et al., 2014). Talreja et al. synthesized ethylene diamine incorporated CNFs (EDA-CNFs) using the chemical vapor deposition process (CVD). Initially, the phenolic beads were synthesized separately and CNFs grown using the CVD process. Next, synthesized CNFs were functionalized using EDA to produce EDA-CNFs for the removal of salicylic acid from water. The data suggested that the prepared EDA-CNFs effectively remove salicylic acid from water with high sorption ability (~682 mg/g) (Talreja et al., 2016). Abega et al. synthesized inorganic/organic nanohybrid of CNT and TiO_2. The composite was synthesized using the surface oxidation of Ti3+ ions method, a modified version of the sol-gel method. The synthesized composite was further applied for the degradation of methyl orange. The synthesized material shows 87% degradation within 100 min of light irradiation (Abega et al., 2019). Kang et al. synthesized Co/Ni loaded CNTs doped with nitrogen for the degradation of organic pollutants, including naproxen, sulfachlorpyridazine, phenol, methylene blue, and methyl orange. The synthesized catalyst demonstrates excellent degradation ability toward organic pollutants (Kang et al., 2019). $CoFe_2O_4$ doped 3D graphene aerogels were synthesized by Li et al. Hydrothermal method was used to synthesize $CoFe_2O_4$ loaded graphene aerogels. The designed photocatalyst demonstrates excellent degradation of benzotriazole (Li et al., 2020). Khalil et al. synthesized porous graphene using a modified method. To create the porosity, synthesized graphene oxide was kept under thermal treatment and further tested for the degradation of several pharmaceuticals compounds, including atenolol, carbamazepine, ciprofloxacin, diclofenac, gemfibrozil, and ibuprofen. Maximum degradation was found for all pharmaceutical compounds (Khalil et al., 2021). Another study utilizes activated carbon as support for TiO_2. The material was synthesized using a sol-gel process and applied as a photocatalyst to degrade 2-methylisoborneol (2-MIB) (Guo et al., 2021). Table 13.1 summarizes the different materials prepared with different synthesis aspects for the removal of pharmaceutical compounds. The data suggested that the synthesis process might affect the productivity or yield of the materials. Interestingly, the removal of pharmaceutical compounds is mainly due to the materials' surface texture and chemical composition. Usually, the metal or non-metal doped CNMs have proved excellent candidates for the degradation of several water born contaminants. These studies report that CNMs can act as excellent support material as they can provide mechanical strength, with charge distribution and large pore size distribution. Additionally, it is important to mention here that the porous texture of the materials might be beneficial for removal of ECs, in terms of more active sites, improved exposure time, and high surface area. With the help of synthesis processes like carbonization and activation, we can easily improve the surface area and active site of the materials, thereby obtaining high removal/degradation efficiency (Fig. 13.1).

13.3.2 Removal of Endocrine-Disrupting Compound

Endocrine-disrupting compounds affect the binding, synthesis, transport, and secretion of hormones within the human body. The endocrine-disrupting compounds are present everywhere in the environment, like air, soil, and water. They modulate metabolism and synthesis of neuro-hormones/peptides/transmitters, resulting in severe complications such as altered development, altered reproduction, and behavioral and hormonal changes. Therefore, there is a need to remove endocrine-disrupting compounds from water bodies. Numerous CNMs have been used to treat/remove endocrine-disrupting compounds from water (Jung et al., 2015; Komesli et al., 2015; Diamanti-Kandarakis et al., 2009; Kumar et al., 2020; Ojha et al., 2021;

Table 13.1 Different CBMs for the removal of pharmaceutical compounds from water

S. No.	Materials	Process	Removal/ degradation efficiency	References
1.	Ni-CB	Carbonization	~300 mg/g	Saraswat et al. (2012)
2.	Fly ash	Acetaminophen	~256 mg/g	Galhetas et al. (2014)
3.	EDA-CNFs	CVD	~682 mg/g	Talreja et al. (2016)
4.	3D-r-GO/ TiO_2 aerogel	Hydrothermal	90% within 90 min	Nawaz et al. (2017)
5.	N-doped-nanoporous Carbon Nanospheroids	Hydrothermal	99.82%	Kundu et al. (2018)
6.	Magnetic-Ni-Co-N-CNTs	Pyrolysis	99, 100, 89, 85, and 78% for MB, MO, naproxen, SCP and phenol.	Kang et al. (2019)
7.	CNT-TiO_2	In-surface oxidation	87% removal within 100 min	Abega et al. (2019)
8.	$CoFe_2O_4$@3D-graphene	Hydrothermal	100% within 180 min	Li et al. (2020)
9.	Boron/N-graphene aerogels	Hydrothermal	88% removal within 3 h	Jiang et al. (2020)
10.	TiO_2-PAC	Sol-Gel	97.8% removal within a 180 min	Guo et al. (2021)
11.	Ni-Carbon micro/ nanoparticles	Suspension polymerization followed by calcination/ Activation	300 mg/g at 300 ppm	Barhoum et al. (2021)
12.	Porous graphene	GO synthesized by modified Hummers' method followed by exfoliation, reduction, and thermal treatment.	90% removal within 180 min	Khalil et al. (2021)

González-González et al., 2022). For example, Kiran Kumar and Venkata Mohan use MW-CNTs to effectively remove natural and synthetic endocrine disruptor compounds like estriol (natural) and 17a-ethynylestradiol (synthetic) from water. The data suggested that the MW-CNTs effectively remove both natural and synthetic compounds from water (Kiran Kumar & Venkata Mohan, 2012). Sinha and Jana et al. synthesized graphene and Fe (G-Fe) based composite to remove 1-naphthol, dibutyl phthalate, bisphenol A, and atrazine from water. The data suggested that the remarkable sorption ability of the G-Fe against 1-naphthol, bisphenol A, dibutyl phthalate, and atrazine were observed at 680, 360, 200, and 200 mg/g, respectively. The high surface area of the G-Fe materials is mainly responsible for the higher adsorption ability (Sinha & Jana, 2013). Zhang et al. synthesized graphene oxide (GO) incorporated polyethyleneimine (PEI) (GO-PEI) based

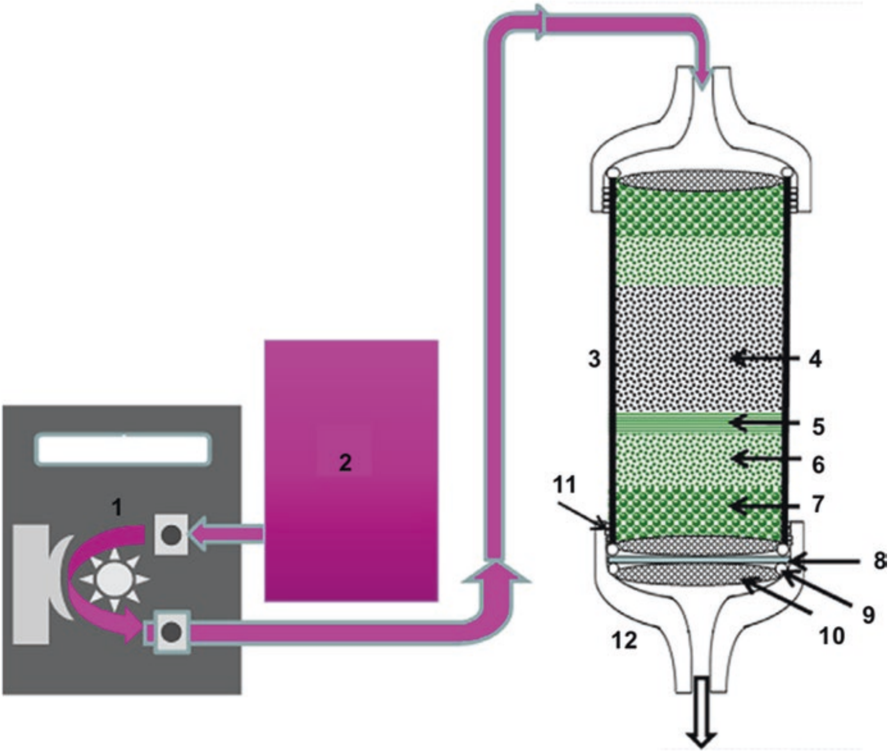

Fig. 13.1 Schematic illustration of the Ni-CB column and removal of vitamin B_{12}. (1). Pump, (2) vitamin B_{12}, (3) column, (4) Ni-CB (200 mm), (5) Ni-CB (0.05 mm), (6) Ni-CB (0.5 mm), (7) Ni-CB (0.8), (8) RO-membrane, (9) O-ring, (10) SS mesh, (11) thread-fitting, and (12) End-stub. (The image was taken with permission Saraswat et al. (2012))

composite material for the degradation of bisphenol A from water. The data suggested that the prepared GO-PEI-based composite materials effectively remove bisphenol A from water with ~20.95 mg/g. The prepared GO-PEI-based composite materials show good adsorption ability with high selectivity of the contaminants (Zhang et al., 2015b). Gupta et al. synthesized GO incorporated β-cyclodextrin (GO-Cyc) to remove bisphenol A from water. The data suggested that the prepared GO-Cyc-based composite materials show effective adsorption ability with ~373 mg/g within 10 min of exposure attributed to rapid adsorption ability (Gupta et al., 2017). Alothman et al. synthesized MW-CNTs using the CVD process to remove 4-Tert-octylphenol from water. The data suggested that the prepared MW-CNTs have the potential ability to remove 4-Tert-octylphenol from water with 94% efficiency within 70 min of exposure time. Moreover, prepared materials are effective, cost-effective, rapid, and reusable (Alothman et al., 2019). Hetero atom doped graphene was synthesized using a one-step hydrothermal method. Boron and nitrogen were considered the best dopants as they can promote charge transfer by reducing the rate of recombination. The synthesized material was used for the

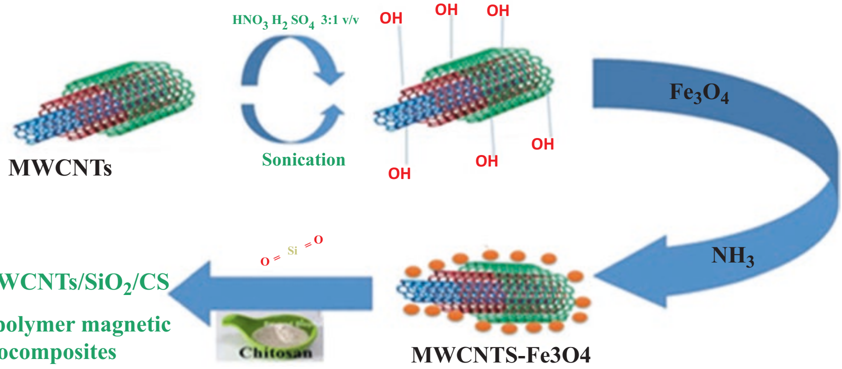

Fig. 13.2 Schematic representation of the synthesis of M-MW-CNTs-P materials. (The image was taken with permission Mohammadi et al. (2020))

photodegradation of bisphenol A (Jiang et al., 2020). Mohammadi et al. synthesized magnetic MW-CNTs incorporated with polymer (M-MW-CNTs-P) for the removal of bisphenol. Figure 13.2 shows the schematic illustration of the synthesis of M-MW-CNTs-P materials. The data suggested that the prepared M-MW-CNTs-P shows the effective removal of bisphenol with approximately 46.2 mg/g adsorption ability at around 76 min. Moreover, M-MW-CNTs-P shows spontaneous and exothermic reactions (Mohammadi et al., 2020).

N'Diaye et al. synthesized magnetic graphene oxide (GO) incorporated polyethyleneimine-Fe-nanoparticles (GO/PEI/Fe) based composite materials to remove bisphenol, progesterone, and norethisterone from water. Figure 13.3 shows the photographic and SEM images of the GO/PEI/Fe-based composite materials. The data suggested that the prepared GO/PEI/Fe-based composite materials effectively remove all endocrine receptors like bisphenol, progesterone, and norethisterone from water with 68, 49, and 80%, respectively. The high adsorption ability of the composite materials is mainly due to porous texture and high surface area (N'Diaye et al., 2020).

Hernández-Abreu et al. synthesized xerogel and lignin-based activated carbon (L-AC) to remove bisphenol from water. The data suggested that both materials show high adsorption ability. Moreover, L-AC shows higher adsorption ability (~220 mg/g), whereas xerogel shows only ~78 mg/g. The higher adsorption ability or removal of bisphenol is mainly due to the porous texture (Hernández-Abreu et al., 2021). Wang et al. synthesized graphene-based materials like GO, reduced-GO (r-GO), and commercial graphene (CG) to remove 17α-ethinyl estradiol and bisphenol A. The data suggested that all materials effectively remove both endocrine disruption compounds. Moreover, r-GO shows higher adsorption ability due to the increased hydrophobicity and micropores. Interestingly, surfactants adversely affect adsorption because of their blocking of the pores (Wang et al., 2021a). Table 13.2 summarizes different CNMs and their application for removing endocrine disruptor compounds. The data suggested that the CBNs effectively remove

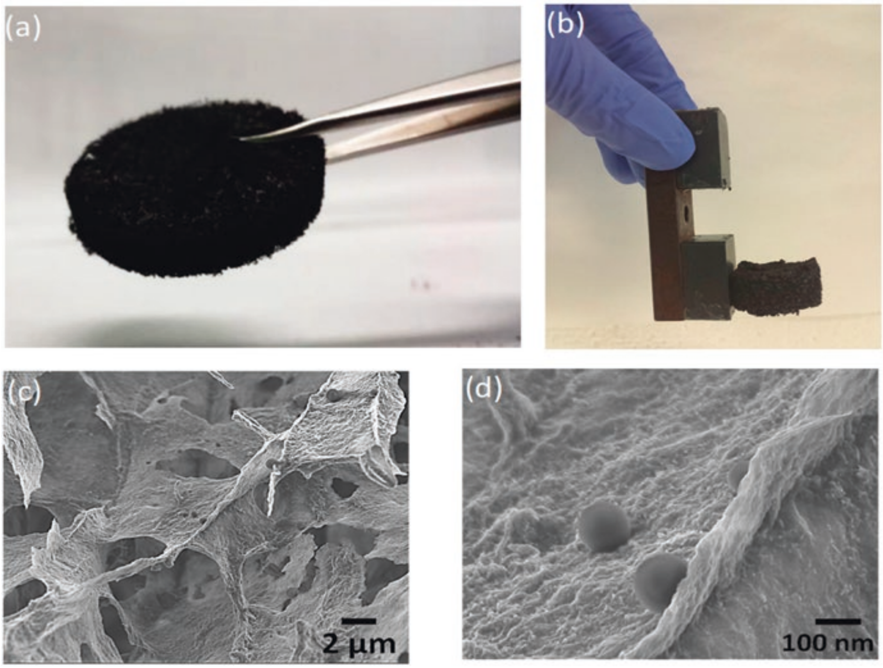

Fig. 13.3 Photographic and SEM images of the GO/PEI/Fe-based composite materials. (The image was taken with the permission (N'Diaye et al., 2020) of the Creative Commons Attribution (CC BY) license)

endocrine disruptor compounds from water. Interestingly, surface medication by metals, polymers, and chemicals significantly enhances the removal efficiency. Moreover, high surface area/high porosity considerably improves the removal efficiency of the endocrine disruptor compounds.

13.3.3 Removal of Personal Care Product

The water contamination increased day-by-day with the development in the economy, population, and industrial revolution due to increment in the production of waste that leads to environmental pollution. Personal care products/ECs include pesticides, insecticides, musk, detergents, sunscreen filters, triclosan (lipid-based antimicrobial agents), and antiseptics. Personal care products have been in the limelight for a few years now because of their presence in the environment and their possible adverse effects on humans and the environment. It is important to mention here that removal of these personal care products from wastewater is quite complex because of extremely low concentrations and is one of the greatest challenges today (Oluwole et al., 2020; Ternes et al., 2015; Ebele et al., 2017; Blair et al., 2015;

Table 13.2 Different CNMs and their application for removing endocrine disruptor compounds from water

S. No.	Materials	Endocrine disruptor compounds	Removal/ degradation efficiency	References
1.	MW-CNTs	estriol (natural), and 17a-ethynylestradiol (synthetic)	–	Kiran Kumar and Venkata Mohan (2012)
2.	G-Fe	1-naphthol, bisphenol A, dibutyl phthalate, and atrazine	680, 360, 200, and 200 mg/g	Sinha and Jana (2013)
3	GO-PEI	Bisphenol A	20.95 mg/g	Zhang et al. (2015b)
4.	GO-Cyc	Bisphenol A	373 mg/g	Gupta et al. (2017)
5.	C_{60} modified ZnAlTi-LDO	Bisphenol A	80%	Ju et al. (2017)
6.	MW-CNTs	4-Tert-octylphenol	94%	Alothman et al. (2019)
7.	M-MW-CNTs-P	Bisphenol	46.2 mg/g	Mohammadi et al. (2020)
8.	GO/PEI/Fe	Bisphenol, progesterone, and norethisterone	68, 49, and 80%	N'diaye et al. (2020)
9.	Xerogel and L-AC	Bisphenol	~78 and 220 mg/g	Hernández-Abreu et al. (2021)
10.	GO, r-GO, and CG	Bisphenol A	–	Wang et al. (2021a)

Kasprzyk-Hordern et al., 2008; Yang et al., 2017; Tijani et al., 2016; Boxall et al., 2012; Wang & Wang, 2016). In this context, nanomaterials, especially CNMs have the potential ability to easily remove personal care products from water. Numerous CNMs and their composite materials have been used to remove personal care products. For example, Zhang et al. synthesized cellulose-graphene-based composite (CGC) to remove triazine from water. The data suggested that the prepared CGC-based composite materials effectively remove triazine pesticides (~75% removal efficiency) from water (Zhang et al., 2015a). Liu et al. synthesized magnetic graphene oxide (M-GO) to remove disinfection by-products. The data suggested that the prepared M-GO effectively removes disinfection by-products with a range of 78–98% efficiency. Moreover, incorporating Fe significantly improves the removal efficiency (Liu et al., 2015). Lazarević-Pašti et al. used GO, single graphene layer (GS), and graphene nanoplate (GNP) to remove dimethoate and chlorpyrifos from water. The data suggested that the GNP has a high sorption ability (0 and ~140 mg/g) for dimethoate and chlorpyrifos. The GS has 37 and 79 mg/g sorption ability of dimethoate and chlorpyrifos, respectively, whereas GO has least 5.2 and 0 mg/g sorption ability of dimethoate and chlorpyrifos, respectively. Moreover, the functionalization of graphene improved the sorption ability of the pesticide (Lazarević-Pašti et al., 2018). Tong et al. synthesized MgO and biochar from corn stalk for the removal of tricyclazole, malathion, and isoprothiolane pesticide. The data suggested that the biochar has the highest removal efficiency (1260–2778 mg/Kg) than MgO

Table 13.3 Different CNMs and their composite for the removal of personal care products

S. No.	Materials	Personal care products	Removal efficiency	References
1.	CGC	Triazine	85%	Zhang et al. (2015a)
2.	M-GO	Disinfection by product	78–98%	Liu et al. (2015)
3.	GO	Dimethoate and chlorpyrifos	5.2 and 0 mg/g	Lazarević-Pašti et al. (2018)
4.	GS	Dimethoate and chlorpyrifos	37 and 79 mg/g	Lazarević-Pašti et al. (2018)
5.	GNP	Dimethoate and chlorpyrifos	0 and 140 mg/g	Lazarević-Pašti et al. (2018)
6.	MgO and Biochar	tricyclazole, malathion, and isoprothiolane	6–64, 98–100, and 47–79%	Tong et al. (2019)
7.	GO composite	Caffein, and triclosan	14.8, and 14.5 mg/g	Delhiraja et al. (2019)
8.	Cyc-ACG	Cymoxanil and imidacloprid	~75%	Utzeri and Verissimo (2021)
9.	GO	Carbaryl	–	Wang et al. (2021b)

(28–2193 mg/Kg) because of the surface functional group on the biochar. Moreover, malathion has maximum removal efficiency due to chemisorption (Tong et al., 2019). Delhiraja et al. synthesized GO-based composite material to remove pharmaceutical compounds and personal care products like caffeine and triclosan. The data suggested that the prepared GO-based composer materials effectively removed caffeine and triclosan with 14.8 and 14.5 mg/g, respectively (Delhiraja et al., 2019). Utzeri and Verissimo et al. synthesized poly-β-cyclodextrin incorporated activated carbon gel (Cyc-ACG) to remove pesticides from water. The data suggested that the approximately 5–10% activated carbon-based Cyc-ACG shows high removal efficiency of ~75% from water (Utzeri & Verissimo, 2021). Wang et al. synthesized GO for the removal of carbaryl from water. The data suggested that the GO effectively removes carbaryl from water (Wang et al., 2021b). Table 13.3 summarizes the different CNMs and their composite for removing personal care products. The data suggested that the CNMs effectively remove personal care products from water. Moreover, the incorporation of metal/surface functional group/polymers significantly improved the efficiency of the materials.

13.4 Conclusion

With the help of NMs, the efficiency of these processes might be significantly improved because these processes mainly depend on the materials. The changes in the characteristics of the material and the processes' removal/degradation/sorption ability dramatically increased. Moreover, NMs show superior activity compared to bulky materials due to improved surface-to-volume ratio and reactive sites. Interestingly, the removal of ECs was mainly due to the materials' surface texture and chemical composition. Usually, the metal or nonmetal doped CNMs have proved excellent candidates for the degradation of several water born contaminants. This study reports that CNMs/CNMs-based composite can act as excellent support material as they can provide mechanical strength, with charge distribution and large pore size distribution. Additionally, it is important to mention here that the materials' porous texture might be beneficial for removing ECs, in terms of more active sites, improved exposure time, and high surface area. With the help of synthesis processes like carbonization and activation, we can easily improve the materials' surface area and active site, thereby achieving high removal/degradation efficiency.

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