

Review Article

Insights on the Dynamics and Toxicity of Nanoparticles in Environmental Matrices

T. Devasena ¹, **B. Iffath**,¹ **R. Renjith Kumar**,¹ **Natarajan Muninathan**,²
Kuppusamy Baskaran,² **T. Srinivasan**,³ and **Shani T. John** ⁴

¹Centre for Nanoscience and Technology, A.C. Tech Campus, Anna University, Chennai 600025, Tamil Nadu, India

²Central Research Laboratory, Meenakshi Medical College Hospital and Research Institute, Meenakshi Academy of Higher Education and Research, Kanchipuram, Tamil Nadu, India

³Centre for Research and Development, Department of Microbiology, Hindustan College of Arts and Science, Chennai, Tamil Nadu, India

⁴Department of Biology, School of Natural Science, Madawalabu University, P.O. Box 247, Oromiya Region, Bale Robe, Ethiopia

Correspondence should be addressed to T. Devasena; tdevasenabio@gmail.com and Shani T. John; shani.dip@gmail.com

Received 29 April 2022; Revised 21 June 2022; Accepted 29 June 2022; Published 31 July 2022

Academic Editor: Senthil Rethinam

Copyright © 2022 T. Devasena et al. This is an open access article distributed under the Creative Commons Attribution License, which permits unrestricted use, distribution, and reproduction in any medium, provided the original work is properly cited.

The manufacturing rate of nanoparticles (10–100 nm) is steadily increasing due to their extensive applications in the fabrication of nanoproducts related to pharmaceuticals, cosmetics, medical devices, paints and pigments, energy storage etc. An increase in research related to nanotechnology is also a cause for the production and disposal of nanomaterials at the lab scale. As a result, contamination of environmental matrices with nanoparticles becomes inevitable, and the understanding of the risk of nanotoxicology is getting larger attention. In this context, focusing on the environmental hazards is essential. Hence, this manuscript aims to review the toxic effects of nanoparticles on soil, water, aquatic, and terrestrial organisms. The effects of toxicity on vertebrates, invertebrates, and plants and the source of exposure, environmental and biological dynamics, and the adverse effects of some nanoparticles are discussed.

1. Introduction

There is a steady upsurge in the manufacturing rate of nanoparticles (10–100 nm) owing to their widespread applications in the fabrication of nanomaterial-based products related to pharmaceuticals, cosmetics, medical devices, paints and pigments, energy storage etc. Metallic nanoparticles were used as antibacterial agents due to their ability to induce reactive oxygen species formation and cellular damage [1]. Nanoparticles possess therapeutic applications due to enhanced permeation and retention effect [2, 3] and sensing effect due to magnetic and optical properties [4, 5]. Food industries are progressing using nanoparticles which form a barrier against gases, humidity, and other factors that could alter and reduce food stability [6]. Iron oxide NPs are used in biofortification, while $\text{Cu}(\text{OH})_2$ nanoparticles are used as an eco-friendly pesticide in the field of agriculture

[7, 8]. Nowadays, nanoparticles have been used to a greater extent in sunscreen and sports equipment due to their unique properties [9, 10]. Apart from the abovementioned applications, nanoparticles are used in bone reconstruction [11], car tire reinforcements, speakers' heat transfer [12], etc. The production rate of different types of nanoparticles is in the following order: titanium oxide and silicon dioxide nanoparticles > cerium oxide nanoparticles > zinc oxide nanoparticles > carbon nanotubes > silver nanoparticles [13–15]. An increase in the production rate of nanoparticles enhances their release of nanoparticles in our environment (Figure 1). As a result, the following aspects have become the important factors of nanotoxicology [16]:

- (1) The potential source of exposure.
- (2) Various analytical methods to quantify the environmental concentrations.

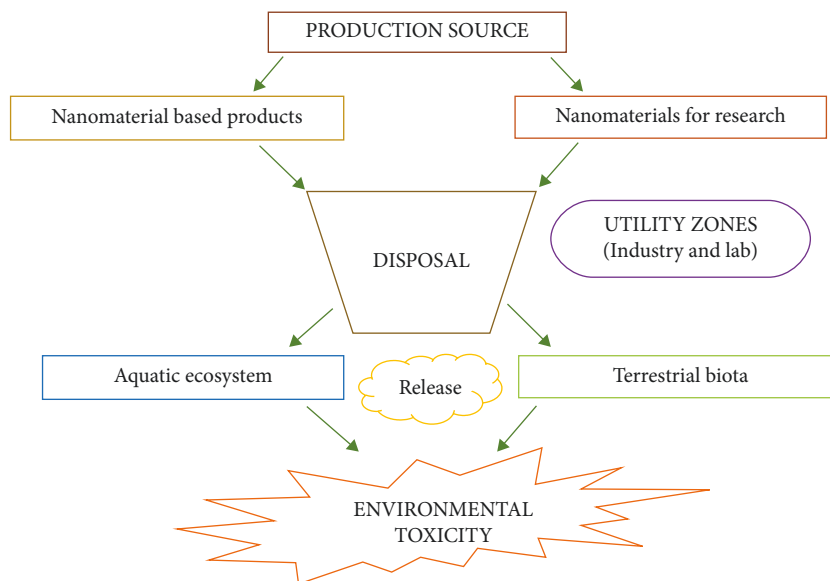


FIGURE 1: Sources of nanoparticles and their environmental entry.

- (3) The fate of nanoparticles in the environment (aquatic and terrestrial living beings).
- (4) Adverse effects of nanoparticles on biological systems such as bioaccumulation and toxicity.
- (5) Possible nanotechnological strategies to combat nanotoxicity.

The environmental effects of nanoparticles are mainly attributed to their small size, high surface area, high surface reactivity, high aspect ratio, and diverse morphology. Studies reveal that partially functionalised metal-based nanoparticles and carbon-based nanoparticles are present in the environment [17–19]. Topics such as nanometrology, dosimetry, transformations, persistence, and bioaccumulation of nanomaterials in the complex environmental media have become very important facets of environmental nanotoxicology studies for the current industrial era [20–22]. Hence, our review systematically focuses on the source of exposure, environmental and biological dynamics, and the adverse effects of nanoparticles, whose production rate is constantly on the rise.

2. Sources of Nanoparticles Contributing to Environmental Toxicity

The environment of the living system such as plants, aquatic organisms, and terrestrial organisms including humans are exposed to nanoparticle candidates primarily via three sources [23–25], namely production source [26], application sites, and disposal sites [27]. The emission pathway and the release pattern usually vary for every type of nanoparticles. Titanium dioxide (TiO_2), zinc oxide (ZnO), and silver (Ag) nanoparticles are regarded as the major OECD (Organization for Economic Co-operation and Development) priority materials. Hence, nanotoxicologists and ecotoxicologists are currently focusing on the toxicological properties of these specific OECD relevant materials. TiO_2

and ZnO nanoparticles are major constituents of cosmetics that are ultimately released into the water. These nanoparticles are mainly released into the environment during water treatment process and hence they mainly accumulate in the soil [28, 29]. TiO_2 nanoparticles are reported to induce size-dependent toxicity to algal species. ZnO and silver nanoparticles were however reported to undergo ready dissolution and their ions were found to be toxic to the aquatic organisms [30].

Aluminium oxide/alumina (Al_2O_3) nanoparticles are among the most abundantly produced chemical in nano-sized particles, estimated to account for approximately 20% of the world market of nanoparticles. Alumina nanoparticles' production rises due to their application in catalysis, structural ceramics for reinforcements, polymer modification, functionalisation of textiles, heat transfer fluids, and wastewater treatment. In addition, Al_2O_3 nanoparticles have shown wide biological applications in biosensors, bio-filtration, drug delivery, and antigen delivery for immunisation purposes. There is evidence that exposure to aluminium may also contribute to an increase in oxidative stress, inflammatory events, and/or the breakdown of the blood-brain barrier (BBB). Al_2O_3 nanoparticles prepared by using micro-emulsion methods and conventional sintering processes were reported to decrease the viability of various cell lines (VERO, HEP 2, A549, and MDA MB 231). Further, the aluminium oxide nanoparticles induced bacterial cell death. The attachment of alumina nanoparticles to the surface of the cell membrane and subsequent disturbance in cellular permeability and respiration were proposed mechanism of action [31].

Carbon nanotubes (CNTs) are widely used in the aerospace, automotive, and electronics industries because of their stability and enhanced metallic and electrical properties. CNTs are also being investigated for biomedical applications such as drug delivery systems and biosensors. CNTs are well known for oxidative stress, inflammation, apoptosis, pulmonary inflammation, fibrosis, and granuloma in the lungs.

So, it is essential to consider the chronic toxicity of CNTs before using them for various biomedical applications. The toxicity of CNTs has already been reviewed [32]. Time-dependent pulmonary toxicity due to inhalation of multi-walled carbon nanotubes (MWCNTs) was evaluated by the bronchoalveolar lavage fluid (BALF) and histopathological analysis. MWCNTs increased cell count, neutrophils, lymphocytes, lactate dehydrogenase, alkaline phosphatase, protein, and cytokines (tumor necrosis factor- α (TNF- α) and interleukin 4 (IL-4)) with a contaminant decrease in the cell viability and alveolar macrophage count BALF on days 1, 7, and 14 postexposure, when compared to control rats. Histopathological analysis revealed inflammation, fibrosis, and granuloma in the lungs of MWCNT-treated rats on days 7 and 14 postexposure. Thus, it could be inferred that MWCNT induces inflammation, fibrosis, and granuloma characterised by the progressive elevation of TNF- α and IL-4. MWCNT accumulates in the lungs and tracheobronchial lymph nodes (TBLN) [33].

Coal is the main fuel source widely used in power plants for electricity generation, due to its abundance and low cost. Combustion of coal produces coal fly ashes (CFAs) containing toxic constituents such as metals, polycyclic aromatic hydrocarbons, and silica. The worldwide annual production of CFA is estimated at around 300–350 million tonnes through the combustion of 3,000 million tonnes of coal [34]. As revealed by characterisation techniques such as EDAX, atomic force microscopy (AFM), X-ray diffraction, and scanning electron microscopy, CFA consists of nanoparticles of around 50 nm, so-called CFA-Nps. CFA-Nps inhibit cellular metabolism in a dose-dependent manner at concentrations varying from 13 to 800 micrograms per ml. After 48-h exposure, the Hep2, A549, and HepG2 cell lines were found to be more sensitive to CFA-NPs at varying levels. The IC_{50} value of CFA-NP-treated Hep2, A549, and HepG2 cells was found to be 397, 243, and 259 micrograms per ml, respectively [35].

The rapid increase in motor vehicle usage is contributing to high levels of urban air pollution which constitute the main source of fine and ultrafine particles, having a serious impact on our urban air quality and public health. The toxicity of vehicle exhaust nanoparticles was reviewed in which diesel exhaust nanoparticles (DENPs) and petrol exhaust nanoparticles (PENPs) were identified and characterised in vehicle exhaust samples by Durga et al. [36]. These samples were reported to be toxic to monolayer culture of various cell lines (HT29, VER, HEP2, A459, and MDA MB23) as indicated by a decrease in cell viability [37]. In vitro exposure to PENPs induced significant oxidative stress, together with membrane leakage, lipid peroxidation, cell inflammation, and protein release, all of which may be the reason for cellular toxicity. Thus, it could be stated that PENPs have the potential to induce toxicity via oxidative stress and inflammation [38].

3. Tools for Analysing Nanoparticles in the Environment

Improved nanometrology is imperative for understanding the environmental concentrations of nanomaterials and for

allowing accurate dosimetry in ecotoxicology testing. Suitable metrology for imaging and measuring the accurate concentrations of nanoparticles of environmental relevance is needed to assess the exact toxicological effects in the biological systems of both vertebrates and invertebrates [39]. This would help analysing the dose-response relationship more exactly. Even the currently available techniques have various limitations such as poor sensitivity, deprived resolution, and absence of completely quantitative data when using complex environmental samples. Still, surface characterisation and determining concentration of nanoparticles are essential to understand the fate of nanoparticles in the environment and food chain [40]. So far, there is no direct analytical technique to quantify the concentrations of nanoparticles in the environment. However, measurements can be done with the aid of computational modelling methods [29, 41]. There is also a need to differentiate manufactured nanoparticles from the particles already present in the tissue of the organisms considered for ecotoxicity studies. This is challenging; however, isotopically labelled nanoparticles and mass spectroscopy-based approaches are crucial in analysis and in yielding relevant data [42].

Probabilistic approaches are of greater significance [43]. This method involves the time-dependent flow of specific nanoparticles in technical modules and environmental compartments enabling the measurement of several nanograms to micrograms of nanoparticles in water samples. This method deals with the dynamic input rates, in-use stocks, and the continuous rise in production volumes [44]. This method is highly suitable for titanium oxide and zinc oxide nanoparticles.

Single-particle inductively coupled plasma mass spectrometry (sp-ICP-MS) is an analytical technique used for the quantification of certain nanoparticles in the environment. Inorganic engineered nanoparticles such as metal nanoparticles including gold, silver, and copper can be quantified by sp-ICP-MS [45]. Surface active nanomaterials in soil samples containing more than one element such as core-shell nanocomposites can be analysed using a multi-element technique, for example, sp-ICP-Time of Flight (ToF)-MS [46, 47].

Sizing and quantification of nanoparticles in the environmental media are also possible by using hydrodynamic chromatography coupled with UV-visible, fluorescence, and inductively coupled plasma mass spectrometry detectors [47, 48]. Nanoparticles tracking analysis and hyphenated methods such as field flow fractionation with inductively coupled plasma mass spectrometry (FFF-ICP-MS) and single-particle inductively coupled plasma mass spectrometry (SP-ICP-MS) are analytical tools of the modern era that are highly relevant to nanoecotoxicology studies [49, 50]. Nanoparticle tracking analysis is capable of providing more accurate data than the dynamic light scattering. SP-ICP-MS enables the single and ensemble analysis of particle number and concentration on an ion-specific basis. This technique is in fact used for the analysis of nanomaterials accumulated within biological cells [51, 52].

Electron microscopy imaging of tissue slices is very labour-intensive. However, fluorescent polystyrene microspheres can be used as probes for tracking the cellular uptake

of nanoparticles in the juvenile stages of fish. SC-ICP-MS is used for the direct particle detection in alkali-digested tissue samples of environmental relevance [52, 53].

Targeted Raman spectroscopy has been able to recognise intact metal oxide nanoparticles such as ceria, zinc oxide, and titania in or on the surface of fish gills from waterborne exposures. This is highly useful for studying the bioavailability of nanoparticles in fish gills.[54]. Techniques such as gamma spectrometry (pulse-chase experiments) and autoradiography are useful in investigating the trophic transfer of nanomaterials in addition to uptake and assimilation studies. For example, the uptake, assimilation, and trophic transfer of dietary nano-CeO₂ particles (containing gamma-emitting radioisotope Cerium-141) along a freshwater food chain represented by an alga (*Pseudokirchneriella subcapitata*), a grazing snail (*Potamopyrgus antipodarum*), and a prawn (*Macrobrachium australiense*) were studied by Cresswell et al. [55]. The study however substantiated the rapid elimination of nanoparticles in snail and the prawn suggesting the absence of assimilation in these organisms. Micro-X-ray fluorescence spectrometry has the ability to reveal the presence of metal nanoparticles in the freshwater organisms. Such studies on freshwater Asian clam *Corbicula fluminea* reveals the presence of gold nanoparticles in the gut epithelium [56].

Electron microscopy (SEM and TEM) can be used as a complementary technique for elucidating size and structural information of environmental nanoparticles [57]. TEM is useful for getting a clear image of electron-dense particles such as metallic nanoparticles. TEM images are used to obtain idea on the endocytosis pathway of cellular uptake of silver nanoparticles in aquatic organisms which feed on sediment deposits rich in silver nanoparticles. TEM images and EDAX data can reveal clear electron-dense zones of silver nanoparticles in the apical plasma membrane, endocytic pits, and in endosomes of the *Nereis diversicolor* [58]. Transmission electron microscopy is used for imaging nanoparticles such as carbon nanotubes and copper oxide nanoparticles inside the gut. Zhu et al. and Heinlaan et al. have imaged the nanoparticles in *Daphnia magna*. Single-walled carbon nanotubes and copper oxide nanoparticles were reported to accumulate in the gut lumen and midgut [59, 60].

Coupling electron microscopy with other techniques are also useful in the analysis of environmental nanoparticles in tissue samples. For example, energy-dispersive X-ray spectroscopy (EDAX) and electron energy loss spectroscopy (EELS) are used for the quantification and speciation of nanoparticles in tissue samples [61, 62]. However, even at relatively high concentrations of nanoparticles, the probability of single nanoparticle imaging in an electron micrograph of a cell such as the gill cell of a fish is undoubtedly a challenging task due to possibility of false negative results.

Isotope labelling techniques are useful for determining the bioavailability, bioaccumulation, and their relation to toxicity of nanoparticles. Croteau et al. have reported the importance of isotope labelling strategies for the determination of bioaccumulation and toxicity of copper oxide nanoparticles in freshwater invertebrates following

waterborne and foodborne exposure. It was concluded that diet-borne copper oxide nanoparticles are more deleterious readily than waterborne particles [63]. Tools that are available to characterise nanomaterials in environmental samples are shown in Table 1.

Modelling nanoparticles in the environment was commenced by [64], who presented the first quantitative approach for assessing nanoparticle release and concentrations for environmental media. He described the theoretical basis on the nanoparticle release quantification that opened the field for several subsequent modelling studies. Several algorithms were developed for calculating the predicted environmental concentrations for a series of nanoparticles in water, biosolids, and soils. Due to a virtually complete lack of empirical information on nanoparticle production and use amounts, the calculations were fully based on a hypothetical model input and therefore not further used in evaluation. Mueller and Nowack [29] for the first time used a material flow analysis (MFA) to replace hypothetical calculations. Park et al. [65] employed emission and atmospheric dispersion models for their work on nanosized CeO₂ emissions from its use as a diesel additive. Blaser et al. [66] presented modelled concentrations of Ag originating from the use of biocidal Ag applications including nano-Ag. Koelmans et al. [67] made the first steps in environmental fate modelling for carbon-based nanomaterials in sediments by combining the output of Mueller and Nowack's release model with mass balance calculations involving agglomeration, sedimentation, and burial nanomaterial in the deeper layers of the sediment for the first time. Gottschalk et al. [68] used a probabilistic material flow analysis (PMFA) approach [69] built on Monte Carlo (MC) computer simulations for assessing predicted environmental concentrations of five nanomaterials (TiO₂, ZnO, Ag, CNT, and fullerenes) in water, sediments, biosolids, soils, and air. Musee [70] used a deterministic and scenario-based MFA to calculate nanomaterial emissions from cosmetic products into the water and terrestrial environments. Johnson et al. [71] estimated the predicted environmental concentrations of sunscreen TiO₂ for soils by basing their PEC modelling on their own measured concentrations in biosolids. Arvidsson et al. [72] proposed a particle flow analysis approach (PFA) to assess anthropogenic nanomaterial release into the environment.

4. Environmental Dynamics of Nanoparticles

The rapid increase in the environmental concentration of nanoparticles, high bioavailability [73], and novel behaviour of nanoscale materials in the environment and ecosystem are responsible for the deleterious biological effects of nanoparticles [24, 74, 75]. In addition, a rapid hike in the production rate of nanoparticles is estimated to rise further thereby increasing their eventual toxicity in the environment [76]. Physiochemical properties of nanomaterials such as size, shape, specific surface area, elemental composition, surface functionalisation, and crystalline structure are also the major determinants of their environmental dynamics which leads to eco-nanotoxicity and bio-nanotoxicity [77, 78]. Once the nanomaterials are released into the

TABLE 1: Techniques used for the characterisation of nanomaterials in environmental samples.

| S. no | Techniques | Applications in environmental nanotoxicology assessment |
|-------|--|--|
| 1 | Dynamic light scattering | Gives an idea of the particle sizes in the whole dispersion |
| 2 | Differential centrifugal sedimentation | To generate fractions containing different particle sizes |
| 3 | Size exclusion chromatography | Chromatographic fractions are generated according to molecular weight, enabling plots of size distribution |
| 4 | Field flow fractionation technique | High-resolution particle size distributions and separation of subfractions |
| 5 | Electrophoretic mobility | To determine the surface properties of nanoparticles that is net surface charge |
| 6 | N ₂ absorption (BET analysis) | For measuring the specific surface area of particles |
| 7 | Gravimetry | To filter the environmental samples |
| 8 | Turbidimetry | Particle concentration can be estimated |

environment, their tendency to aggregate, surface binding ability, potential to release toxic metal ions, capacity to passivate, or capability to interact with various environmental or biological constituents such as humic substances, (muco)polysaccharides, and cellular debris will change. As a result, their bioavailability and toxicity will also change [79–81].

Once the nanoparticles enter into the environment, they undergo dynamic transformation. Apart from the properties of nanoparticles, the environmental parameters such as pH, ionic strength, organic and inorganic colloids, temperature, etc., can also regulate the transformation process [82]. Three major transformation processes, namely physical, chemical, and biological transformations, can clearly explain the fate of nanoparticles in the environment [83].

- (1) Physical transformation includes aggregation, agglomeration, sedimentation, and deposition (in porous media).
- (2) Chemical transformation includes dissolution and subsequent speciation changes, redox reactions (oxidation and sulfidation), photochemical reactions, and corona formation.
- (3) Biological transformations that encompass biodegradation and biomodification.

Various factors such as the pH, the ionic strength, the presence of divalent ions, the type/concentration of organic matter, and the quantity of the engineered nanoparticles strongly influence the aggregation process [84–86]. However, the presence of natural organic matters such as humic acid and fulvic acid are known to block aggregation [87]. Homoaggregation was reported to be insignificant at realistic environmental concentrations (<1 µg/L) and relevant timescales, while the heteroaggregation was quantitatively significant due to the higher concentrations of natural colloids in the environmental media. Natural organic matter is capable of replacing the surface coating of the nanoparticles and subsequently causing electrosteric repulsion and stability [88]. Merrifield et al. have also reported that the aggregation kinetics are strongly related to the initial concentration (in particular the number concentration) of the dispersed nanoparticles [89]. Measurement of the number concentration of nanoparticles and the nanoparticle mass (size) using SP-ICP-MS clearly reveals that the aggregation

and the dissolution of nanoparticles depend on the concentration and the mass [62]. It was also reported that the aggregation of nanoparticles can increase the accessibility of the particles to the biomass or the ingestion rate, both of which raise the bioaccumulation [63, 90]. In marine water, nanoparticles are sterically stabilised. This leads to polymer entanglement and bridging thereby promoting aggregation [91, 92]. In the ecotoxicology community, adsorption phenomena involving the binding of metal on fish gills is influenced by physical factors such as the net charge of the surface, ionic mobility of the counterions in the surrounding medium (which in turn is partly defined by charge density and the hydrated ionic radius of the ion), and competition with other ions in the medium such as H⁺ [93]. Thus, it could be understood that the surface charge of the nanomaterials alters their ecotoxicity.

Quik et al. have demonstrated that the natural organic matters (colloids) are responsible for the sedimentation of nanoparticles in freshwater samples [94]. Quik et al. have also assessed the exposure of aquatic organisms to engineered nanoparticles and reported first-order kinetics for both sedimentation and dissolution [95].

Dissolution refers to the release of specific ions from the parent nanomaterials which in turn cause deleterious effects to the cells. Bacteriolysis is an important example of dissolution-based effect. Silver nanomaterials are iconic in releasing ions for killing bacteria which is responsible for the well-known antibacterial activity of silver nanomaterials. However, other factors such as surface functionalisation, shape, and size of the nanoparticles determine the transport, bioavailability, and the rate and extent of dissolution [96]. The following factors influence the solubility and dissolution of the nanoparticles, which in turn has an effect on their fate and toxicity in the environment: chemical nature of the nanoparticles [97, 98], size of the particles [99], surface coating [100], doping [101], and presence of natural organic matter [89, 102]

Solubility and dissolution are very low for carbon-based nanomaterials and inorganic nanomaterials. Hence, solubility does not contribute to their toxicity. However, for metal nanoparticles such as silver, copper oxide, and quantum dot exert their adverse effects due to their direct biological interaction and their accumulation within the cells. Their toxic effects are determined by the intermediate dissolution and solubility of their ions and particles as well

[103]. Solubility contributes to toxicity for zinc oxide nanoparticles. For carbon-based NMs and many inorganic NMs such as ceria and titania, whose solubility is low, dissolution and solubility become less important.

Sulfidation is a method of chemical transformation that involves surface modification by sulphide group that is available in wastewater treatment effluent and sediments. This process is an oxygen-dependent reaction, more predominant among metal nanoparticles which can alter the particle size, surface charge, and solubility and morphology [104]. Subsequently, the fate and the bioavailability of the metal nanoparticles get altered. [105, 106] For example, sulfidation of silver nanoparticles to silver sulphide reduces the toxicity [107]. Sulfidation of zinc oxide nanoparticles enhances the aggregation by reducing the surface charge [108]. Copper oxide (CuO) nanoparticles undergoes slow sulfidation reaction forming CuS that releases more quantity of copper ions as compared to the precursor nanoparticles resulting in more toxicity to aquatic biota [109, 110]. Silver, iron oxide, and ceria nanoparticles are prone to redox transformations. Photochemical oxidation of nanoparticles was reported to play a role in the bacterial toxicity and hence exploited as an important antibacterial mechanism [111, 112]. Nanoparticles such as carbon nanotubes, fullerene, TiO₂, and ZnO are photoactive which get excited by light and generate reactive oxygen species (superoxide anions, hydroxyl radicals, etc) that are hazardous and toxic to the living organisms [82]. Dissolution, homoaggregation, heteroaggregation, fragmentation, sedimentation size exclusion, straining, deposition, and convective transport are the factors governing the fate of nanoparticles in soil [113]. In the native environment, redox reaction comprises of oxidation-reduction reactions. The coupled process of oxidation and reduction reaction involves the transfer of electrons between the reacting chemical components and this is influenced by environmental conditions such as pH and existence of electron donors (reducing agent) or acceptor (oxidising agent) [114].

Over the last era, many reports have explored the transport of nanomaterials through soil using natural soils, and columns containing inert stationary phases such as quartz beads in columns and appreciated the fate of nanoparticles in terrestrial system [114]. For example, single-walled carbon nanotube through porous media was explored by Jaisi et al. [115]. Fang et al. determined the fate of titania nanoparticles in saturated homogenous soil [116]. Zinc oxide nanoparticles showed higher rate of dissolution [117]. Dissolved zinc oxide nanoparticles transform into a mixture of species such as ZnS, Zn₃(PO₄)₂, Zn-cysteine, Zn-substituted ferrihydrate, and Zn²⁺ adsorbed to mineral surfaces [118, 119]. Carbon-based nanoparticles (carbon nanotubes, fullerene, and graphene) generally do not undergo hydrolysis instead they dissolve via photolysis or microbial decomposition [120, 121]. High concentrations of colloids suspended in the soil pore water enables greater heteroaggregation of nanomaterials in soil, while homoaggregation is comparatively lesser. Consequently, the straining will be increased and diffusion will be increased. However, both the mode of aggregations can be inhibited by

natural organic matter in soil porewater [122]. This concept was reported by studies using ceria nanoparticles [123], engineered silver nanoparticles, gold nanoparticles [124, 125], carbon nanotubes, hematite nanoparticles [126], and titanium dioxide nanoparticles [127].

Carbon-based nanomaterials such as carbon nanotubes get retained in the soil and cause significant straining [128, 129]. Fullerenes interact strongly with organic matter in the soil and get retained [130, 131].

Biodegradation of nanoparticles are more familiar in carbon-based nanomaterials such as graphene oxide, CNTs, and fullerene derivatives. Graphene oxide sheets are susceptible to degradation by neutrophil, myeloperoxidase-induced degradation producing noncytotoxic and non-genotoxic product [132]. Pristine single-walled carbon nanotubes undergo layer-by-layer degradation in the presence of fungal manganese peroxidase and generate carbon dioxide [133]. Similarly, multi-walled carbon nanotubes are also susceptible to horse radish peroxidase-mediated degradation [134]. Basidiomycete fungal enzymes degrade C60 fullerol particles and decompose them [135].

Next to biodegradation, biomodification plays a role in determining the fate of environmental nanoparticles. During biomodification, the natural bioorganic molecules in the environment adsorb to the surface of the nanoparticles forming a new moiety called eco-corona. The sources of these biomolecules in the environment are microbes, plants (phytoplanktons), and animals (zooplanktons) [136]. Biomolecules of corona are made of polysaccharides, proteins, lipids, nucleic acids, etc [137]. Eco-corona may be hard or soft. Hard corona (strong corona) are directly and strongly bound to the nanoparticles, while soft corona (weak corona) is away from the nanoparticle core and bound to the hard core [138]. Eco-corona is a dynamic entity capable of exchanging its constituent biomolecules with the surrounding environmental matrices [139]. Eco-corona is capable of reducing the generation of reactive oxygen species during the interaction of nanoparticles with the zooplankton or the phytoplankton [140] or capable of altering the aggregation and dissolution behaviour of nanoparticles [141, 142]. In the context of toxicity, eco-corona is capable of minimising the toxicity of nanoparticles in various organisms such as algae, plants, and zebra fish [143–145]. As far as the water matrix is concerned, nanomaterials are highly available to the aquatic organisms at >10 m depth than at the surface [146].

5. Biological Fate of Nanoparticles

As mentioned earlier, the nanoparticles undergo dynamic transformation in different environmental matrices. Hence, the living systems in the environment are more susceptible to the effects of transformed nanoparticles (Table 2) rather than the pristine counterparts. Nanomaterials easily influence the aquatic and the terrestrial health probably due to their novel behaviour contributed by:

- (1) high surface reactivity;
- (2) spatial constraint of electronic properties;
- (3) high specific surface area;

TABLE 2: Biological effects of various nanoparticles on aquatic organisms.

| S. no. | Nanoparticles | Biological effects |
|--------|--|--|
| 1 | Silver nanoparticles | Alters membrane permeability and potential. Interacts with thiol groups of sulphur-containing amino acids in the vital enzymes, interact with DNA and prevent replication. |
| 2 | Silicon NPs and fullerene derivatives | Induces membrane damage altering the permeability and transport processes. |
| 3 | Polyvinylpyrrolidone (PVP)-functionalised fullerene | Negatively charged nanomaterials, lysosomal damage, dysfunction, and autophagy. |
| 4 | Single-walled carbon nanotubes | Generates singlet oxygen that can cause lipid peroxidation and other cell damage. |
| 5 | Carbon nanotubes | Blocks ion channels of the cell membrane. |
| 6 | Quantum dots | Targets the gate keepers such as the epithelial barrier of lungs, intestine, and the endothelial barrier of the blood vessels. |
| 7 | Core-shell structures | Targets the nuclear pore complex and histone proteins and impairs cell cycle regulation. |
| 8 | Fullerenes | Induces bioaccumulation and bio persistence (CdSe, CdTe, CdSeTe, ZnSe, InAs, PbSe, CdS, ZnS). |
| 9 | Titanium dioxide nanoparticles (present in sunscreen lotion) | Causes structural deformation, destabilisation, and functional impairment of the DNA molecules. |
| 10 | Tungsten carbide and silver nanoparticles | Generates oxygen radicals that can nick supercoiled DNA. |
| 11 | Iron oxide nanoparticles | Induces genotoxicity. |
| 12 | Tungsten nanofibers and zinc oxide | Interferes with the polymerisation and maturation of actin fibers. Inhibits the cell differentiation and migration. |
| 13 | Fullerene and cerium dioxide | Alters the structure and function of the cell junctions in fish embryo |
| 14 | Quantum dots and silica nanoparticles | Interferes with the cellular energy transduction |
| 15 | Gold nanoparticles | Induces developmental malformation in the zebra fish embryo heart. |
| | | Induces malpigmentation in the eyes of zebrafish leading to altered swimming behaviour. |

- (4) easy transport mechanism in the media;
- (5) effective formation of interface with subcellular organelles of the living cells.

For example, nanoparticles that are tightly constraint in size and oxidation state are capable of giving different algal transcriptomic and metabolomics responses when compared to bulk counterparts [75].

Dynamic interaction of environmental nanoparticles with the biomolecules of the biological organisms results in a new entity called “bio-corona” which is analogous to eco-corona in the environment [137, 147]. The comprehensive details about bio-corona and nano-bio-interfaces are beyond the scope of this review and already discussed elsewhere [148–150]. Biodegradation, macromolecule-mediated formation of nanoparticles, and bio-corona/nano-bio interactions are the principal biological mediated transformations in the environment. The rate and relative significance of these processes depend on the interaction of nanoparticles with extracellular enzymes, extracellular polymeric substances, and microbes. The fate, transport, and potential ecological risks associated with the nanoparticles released into the terrestrial and aquatic ecosystem is determined by the biodegradability of polymer coating material, the type of underlying core/particle and the dominance/population of a particular microbial community [151]. Nevertheless, detailed studies are inadequate about the biodegradation potential of microbes for various organic polymer coatings present on different nanoparticles and

subsequent transformation of nanoparticles under environmentally relevant conditions.

5.1. Aquatic Ecosystem-Invertebrates. Increased usage of engineered nanoparticles (ENPs) in industrial and commercial applications will inevitably end up in the mixing of these nanoparticles’ aquatic environment. Based on a literature review and an overview of toxic effects of ENPs in aquatic system, it could be inferred that invertebrates are sensitive and relevant organisms for assessing the nanoecotoxicological effects of nanoparticles. Experiments conducted using invertebrate model helps one to advance the knowledge on bioavailability, bioaccumulation, and toxicity [152].

Invertebrates show bioaccumulation of nanoparticles. For instance, silver nanoparticles were internalised into the gut epithelial cells of estuarine polychaete *Nereis diversicolor*. In *Daphnia magna* and *Lumbriculus variegatus*, Khan et al. have investigated the accumulation dynamics and acute toxicity of silver nanoparticles. Silver nanoparticles were reported to enter into the cells via endocytosis and accumulate as inorganic granules, organelles, heat denatured proteins, and metallothionein fraction. Metal modelling approaches like biotic ligand model (BLM) and tissue residue approach (TRA) can be used to establish the link between uptake, accumulation, and toxicity of nanoparticles. Khan et al. have studied the bioaccumulation dynamics of silver nanoparticles functionalised with PVP, PEG, and citrate in comparison to dissolved Ag, in *Daphnia magna* and *Lumbriculus variegatus* [58, 153].

Bioaccumulation in invertebrates depends on various material properties such as the composition, size, and solubility and the environmental dynamics (dissolution, aggregation) of the nanomaterials. Nanoparticles of gold, titania, and silica are less bioavailable and less toxic in invertebrates. Nanoparticles of copper oxide, zinc oxide, and silver possess high bioavailability and high toxicity [83]. Lesser the particle size greater the bioaccumulation, probably due to greater bioavailability [154].

Next factor that determines bioaccumulation in invertebrates is the solubility. For example, silver in dissolved form is more bioavailable and showed higher bioaccumulation than particulate form in the estuarine snail *Peringia ulvae* [155], the freshwater snail *Lymnaea stagnalis*, the water flea *Daphnia magna* [156], and the freshwater oligochaete *Lumbriculus variegatus* [157].

Environmental dynamics such as dissolution stage or aggregation stage of the nanoparticles influence the uptake, bioaccumulation, and toxicity of the particles. For example, in aquatic snails, dissolved phase is responsible for the total uptake of silver, copper oxide, and zinc oxide nanoparticles [63, 156–158].

Croteau et al. have proved that aggregation can also affect bioavailability [63, 159] have measured the aggregation dependence of bioavailability of silver, copper oxide, and zinc oxide nanoparticles in the snail *L. stagnalis*. In mussels, aggregation was reported to increase the bioavailability [90].

When engineered nanomaterials enter into the environment, they are likely to mingle with the food or diet of animals and other organisms. These particles integrate with food material by aggregation and sorption processes. Hence, considering the dietary exposure of nanomaterials and its biological effects is equally important to assess ecological and human health risks. Croteau et al. revealed that isotopically labelled foodborne zinc oxide nanoparticles efficiently assimilated in the freshwater snails. Agglomeration does not interfere with the bioaccumulation and toxicity and resulted in lesser food consumption and impaired digestion. Consequently, developmental processes such as growth and reproduction will be affected resulting in population and community changes [159]. Additionally, Croteau et al. have reported the bioaccumulation and toxicity of copper oxide nanoparticles in freshwater invertebrate, *Lymnaea stagnalis*, following waterborne and foodborne exposure. It was concluded that diet-borne copper oxide nanoparticles are more deleterious readily than waterborne particles [63].

In *Daphnia magna*, single-walled carbon nanotubes and copper oxide nanoparticles accumulate in the gut as revealed using TEM imaging [59, 60]. Growth and reproductive ability of freshwater flea, *Daphnia magna* are affected due to titania and zinc oxide nanoparticles in a chronic study of 12 days [160]. Zhao et al. have done acute and chronic toxicity studies on the effects of waterborne and diet-borne silver nanoparticles and its bioaccumulation in a model freshwater cladoceran, *Daphnia magna*. A 48-hour acute study showed high bioaccumulation at 500 $\mu\text{g/L}$. In a chronic study (21 days), waterborne silver nanoparticles significantly inhibited the growth. The chronic effects of silver nanoparticles might

be due to the low food quality of algae contaminated with silver nanoparticles. This study has emphasised the significant of chronic toxicity studies in assessing the environmental risk of nanoparticles [161].

Oysters are an ecologically significant group of filter feeders of marine water. They can serve as a perfect toxicology model for illustrating the potential impacts of nanoparticles to marine organisms. *Crassostrea virginica* is an ideal marine organism for toxicity assessment of nanoparticles. Fullerene (concentration ≥ 100 ppb) was reported to impair embryonic development and cause lysosomal destabilisation associated with reproductive failure. Within 4 hour of exposure, fullerene particles concentrate on the lysosome-rich hepato-pancreatic tissues. Thus, endocytic and lysosomal pathways are the mechanisms by which fullerene exerts its toxic effects. It is also obvious that disturbance in the lysosomal pathway can cause chronic effects to the health and environment [162]. Mean chronic concentration that is hazardous to a given percentage of freshwater species (typically 95%) values were reported to be 0.2 mg/L for fullerene, 4.8 mg/L for CNTs, 1.2 $\mu\text{g/L}$ for silver nanoparticles, and 2 mg/L for titania nanoparticles [163].

Zerovalent nano-iron are capable of rapid dissolution in water such that the released iron particles exceed safe limits of NOECs established for dissolved Fe. This was proved in an acute study conducted using sperms of three key marine invertebrate species *Mytilus galloprovincialis*, *Ciona intestinalis*, and *Psammechinus milliaris*. In vitro fertilisation postexposure resulted in a decrease in success of fertilisation, delayed embryogenesis, and a disruption in the embryo development [164].

From the perspective of predator-prey interactions, a prey item with nanoparticles in the gut lumen will function as a nanomaterial-borne/contaminated food for the predator. In this context, the uptake, consequent assimilation, and the trophic transfer of dietary nano-CeO₂ particles (containing gamma emitting radioisotope Cerium-141) along a freshwater food chain of alga, prawn, and snail were reported by Cresswell et al. [55]. The study asserted the rapid elimination of ceria nanoparticles in snail and the prawn and revealed the absence of assimilation in these organisms.

There are very few reports on ecotoxicology studies on environmentally relevant algae. Algae (*Desmodesmus subspicatus*) on exposure to titanium nanoparticles in growth inhibition test showed that the surface structure/matrix of plant cell walls could act as a surface to grow nanoparticles [165]. Nanoparticles show adsorption to the surfaces of aquatic sediments, algal mats, biofilms, soils, and even the exterior surfaces of organisms [166].

5.2. Aquatic Ecosystem-Vertebrates. Nanoparticle uptake in the fish gills is almost similar to that in the mucus layer and reported to occur primarily via vesicular endocytosis and less likely by diffusion and transporter-mediated processes. The probability of susceptibility of different organs to nanotoxicity are in the order gills > gut > liver > brain. Nanoparticles are excreted via hepato-biliary mechanism or kidney. But the glomerular excretion is not common for all

fishes. The molecular weight cutoff for glomerular filtration is 60 kDa. Hence nanoparticles of few nanometres on par with 60 kDa will undergo glomerular filtration in most but not all the fish varieties. Exceptionally, the glomerular kidneys of stenohaline marine and the sea water-adapted fish in the hyperosmotic environment cannot excrete the nanoparticles. The production of less urine or no urination is the reason for the hyperosmotic environmental species. However, the effect of nanomaterials on the fish varies depending on the fish habitats. For example, marine water shows high salinity or the ionic strength which cause aggregation. Even fishes dwelling in hardwater and benthic species at the sediment interface will be exposed to aggregates of nanoparticles rather than nanoparticulate suspension or dispersed nanoparticles. Soft water (freshwater) on the hand possesses nanoparticles that are stabilised by organic matter such as fulvic acid and humic acid [167]. Fish epithelial surface contains mucus which is analogous to the mucus of other vertebrates. The majority of the mucus layer (95%) is made of water, while mucoproteins (with highly conserved sialic acid, carboxylic acid, and sulphated residues) and electrolytes constitute the remaining proportion. Nanoparticles entangle strongly with mucoproteins, and the binding strength is more for nonspherical morphology. For example, single-walled carbon nanotubes were reported to bind strongly with the gill mucus of trout [93, 125]. From the mucus layer, the nanoparticles move across the epithelial layer of cells to the blood (transcellular movement) and then move across different cells via tight junctions (paracellular movement). Gut epithelia are capable of absorbing the nanoparticles through endocytosis. Albeit gills are the major portal entry, other routes are buccal cavity, olfactory openings, eyes, and urinary/genital openings. Nanoparticles' entry through these routes may damage nerve endings and cause damage to the retina [167].

Overall, the gills, gut, liver, and brain are the possible target organs for the toxic effects of some manufactured NPs in fish. The mechanism of toxicity includes oxidative stress, inflammation, tumor formation in the liver, disturbances in the ion homeostasis, and vascular injury [84]. For example, titania nanoparticles absorbed via gills induce oxidative stress and inflammation in the internal organs with a concomitant rise in the reactive oxygen species and inflammatory factors.

5.3. Effects of Soilborne Nanomaterials on Terrestrial Biota.

Schultz et al. have demonstrated the trans-generational sensitivity in nematodes (*Caenorhabditis elegans*) exposed to silver nanoparticles for 10 generations. Epigenome was thought to be involved in the sensitivity transfer [168]. Carbon-based nanomaterials such as carbon nanotubes, fullerene, and graphene were reported to be nontoxic or less toxic to soil organisms such as earthworm and microbes [169–171]. Carbon nanotubes (double walled) impregnated into soil exerts toxicity to larva of amphibian. Similarly, the larvae of amphibian are prone to the toxicity of iron oxide, titania, zinc oxide, and copper oxide nanomaterials [172, 173]. Oxidised copper nanoparticles may proceed

through food chains. For example, copper nanoparticles move from soil end enter into earthworm, eliciting toxic effects at a concentration $>65 \text{ mg-Cu-kg}^{-1}$ soil [174, 175].

5.4. Target Organs for Water and Diet-Borne Nanomaterials.

Gomes et al. have revealed the bioaccumulation of copper oxide nanoparticles using the mussel *Mytilus galloprovincialis*. Digestive gland is the major site of bioaccumulation of copper oxide nanoparticles. Copper nanoparticles bioaccumulate and exert their toxicity by eliciting oxidative stress. This could be characterised by assaying lipid peroxidation, activities of antioxidant enzymes, and metallothionein levels. Copper nanoparticles are capable of lowering the activity of superoxide dismutase with a concomitant raise in lipid peroxidation and metallothionein content [176].

Nanoparticles manufactured for biomedical and industrial applications are usually functionalised to resist high ionic strength-mediated aggregation. Those particles exhibit high colloidal stability due which they show environmental persistence. As a result, these particles are expected to induce toxicity in aquatic system. The gut epithelium is the target for surface-stabilised metal nanoparticles. For example, *Corbicula fluminea*, a globally distributed clam, is known to uptake and accumulate gold nanoparticles functionalised with bovine serum albumin in the gut [56]. Silver nanoparticles functionalised with citrate molecules are predominant in estuarine sediments which are habitat and food for deposit feeders. *Nereis diversicolor* (deposit feeder) fed with citrate-capped silver nanoparticles were capable of accumulating the particles in their gut epithelium. The nanoparticles appear as electron-rich zones in the apical plasma membrane, endocytic pits, and in endosomes. These zones can be resolved using TEM [125]. Copper nanoparticles induce swelling of goblet cells, necrosis in the mucosa layer, and vacuole formation in the gut [177].

Nanomaterials are capable of eliciting immunotoxicity and neurotoxicity. In fishes, brain histochemistry gets altered due to nanoparticle toxicity in trouts. Copper nanoparticles elicit toxicity in the cell bodies of the neurons in telencephalon and alter the mesencephalon layer integrity, together with cerebral aneurysms [177]. Ramsden et al. have studied the sublethal effects of dietary exposure to titania nanoparticles in juvenile rainbow trout *Oncorhynchus mykiss*. Titania nanoparticles showed bioaccumulation and persistence in brain as evidenced by an inhibition in the activity of Na^+K^+ ATPase, alteration in Cu-Zn homeostasis, without affecting the growth and haematological parameters. In addition, gills and intestine showed enhanced lipid peroxidation [178].

The fact that the liver of aquatic organisms is also a target for nanoparticles was proved by Al-Bairuty et al. in trout models [177]. Hepatitis-like injury and cells with pyknotic nuclei are the toxicity indicators.

Nephrotoxicity of waterborne copper nanoparticles was also reported by Al-Bairuty et al. Trouts treated with copper nanoparticles showed renal tubular epithelial injuries and altered Bowman's space in the kidney. Last but not the least,

copper nanoparticle treatment in trout causes changes in the area of skeletal muscle fibers there by affecting muscular homeostasis [177]. Gills of the rainbow trout exhibits hyperplasia, aneurisms, and necrosis in the secondary lamellae [177].

Olfactory canals are also targets for nanoparticle-induced toxicity. In fishes, the accumulation of nanoparticles in olfactory canals impairs sensation pertaining to alarm chemicals. The antipredator behavioural responses of juvenile rainbow trout (*Oncorhynchus mykiss*) to trout alarm substance were investigated. The effect of nanoparticles includes a delay in freeze response and altered swimming activity. Change in behaviour was accompanied by a significant increase in the ratio of oxidised to reduced glutathione in the brains of fish, specifying some systemic oxidative stress. However, the morphology of olfactory bulb rosette was not affected [179].

External tissues of aquatic organisms are also targets for metal nanoparticles in water media! Sensory lateral line functions get altered in zebra fish (*Danio rerio*) due to the adverse effects of nanoparticles present in the water. The reason is probably the changes in the number of lateral lines neuromasts (LLNs) and rheotaxis behaviour. Copper nanoparticles reduced LLN. Copper nanoparticles and silver nanoparticles were capable of reducing LLN number and rheotaxis behaviour. This results in loss of sensory perception, resulting in altered foraging behaviours, increased susceptibility to predation, and avoidance of fish migrations [180]. This study is a good example for external tissue-mediated toxicity behaviour of metal nanoparticles in aquatic biological system.

5.5. Nano-Bio Interface. Knowledge on the reaction dynamics at the interface of nanomaterials with cells, subcellular organelles, and biomolecules is essential for understanding the toxic effects of nanomaterials. Cellular dynamics is also significant from the perspective of safe use of nanomaterials. Nanoparticles interact with subcellular organelles such as membrane, mitochondria, nucleus, lysosomes etc. Moreover, vital biomolecules such as DNA, proteins, and lipids are also sites for nanoparticle interaction which results in the formation of protein corona, particle wrapping, and free energy. Therefore, it is essential to ensure safe manufacture, usage, and disposal of nanomaterials in the marketplace.

Considering the wide target for nanomaterial interaction and interface formation, certain general mechanism of toxicity has been reported. Possible mechanism of the toxicity of nanomaterials on biological system are as follows:

- (i) disruption of membrane structural integrity;
- (ii) interference with membrane potential;
- (iii) oxidation of proteins;
- (iv) genotoxicity;
- (v) interruption of energy transduction;
- (vi) generation of reactive oxygen species and pro-oxidant status.

Nanomaterials are capable of generating reactive oxygen species that are involved in oxidising fatty acid chains of membranes by interacting with the double bonds. As a result, the integrity, the fluidity, and the membrane potential get altered [181].

Silver nanoparticles bind to biomembranes altering their permeability and potential. Silicon NPs and fullerene derivatives are also capable of inducing membrane damage altering the permeability and transport processes [182]. Silver nanoparticles and negatively charged nanomaterials were able to enter into lysosomes of the cells present in fish gills and intestinal lining by causing loss of membrane integrity, lysosomal damage, dysfunction, and autophagy [183, 184]. Silver nanoparticles interact with thiol groups of sulphur-containing amino acids in the vital enzymes and interact with DNA and prevent replication [185]. Silver nanoparticles are also capable of interacting with DNA and lipids causing their degradation [182]. Polyvinylpyrrolidone (PVP)-functionalised fullerene generates singlet oxygen that can cause lipid peroxidation and other cell damage [186].

Single-walled carbon nanotubes are known to block ion channels of the cell membrane [187]. Nanoparticles are capable of targeting gate keepers such as the epithelial barrier of the lungs, intestine, and the endothelial barrier of the blood vessels. For example, carbon nanotubes were reported to impair the function of biological barriers. Smaller sized particles follow paracellular route of transport while larger particles followed energy-dependent transcellular route of transport into the barrier cells [166, 188].

Iron-sulphur cluster are cofactors for many cellular enzymes. These clusters are targets for nanomaterials. Nanomaterials generate reactive oxygen species which impairs the function of the clusters. Further, the disulphide bridges are formed between the sulphur-containing amino acids of the enzymes leading to an impairment in the structure and the function of the enzymes [189].

Though nanomaterials such as quantum dots are used for DNA labelling and imaging, they induce nick in supercoiled DNA [190]. Quantum dots due to their small size pass through the nuclear pore complex targeting histone proteins that play an important function in cell cycle regulation and tumorigenesis [191]. Core-shell structures are capable of causing bioaccumulation and cause adverse effect due to biopersistence. Semiconductor quantum dots such as CdSe, CdTe, CdSeTe, ZnSe, InAs, or PbSe contain noble or transition metals in their core. Similarly, quantum dots such as CdS or ZnS contain metals in their shell [192, 193].

Fullerenes bind DNA and cause structural deformation, destabilisation, and functional impairment of the DNA molecules and cleavage [194–196]. Titanium dioxide nanoparticles (present in sunscreen lotion) generate oxygen radicals that can nick supercoiled DNA [197]. Tungsten carbide and silver nanoparticles induce genotoxicity (micronuclei formation) in mammalian cells. Mechanism of action includes oxidative stress and interference with microtubules and actin filaments during cell division [198–200].

Iron oxide nanoparticles establish steric interactions with the cytoskeleton network and interfere with the polymerisation and maturation of actin fibers, thereby

inhibiting the cell differentiation and migration [201]. Tungsten nanofibers and zinc oxide nanoparticles enter into the epithelial lining of the hepatopancreas and frog embryo enterocytes causing oxidative stress and alteration in the structure and function of the cell junctions [202, 203].

Engineered nanoparticles such as titania, silver, polystyrene, and carbon nanoparticles were reported to cause mitochondrial dysfunction thereby impairing energy production [204]. Electron transport chain, oxidative phosphorylation, and energy transduction are vital processes of the cell. Nanoparticles interfere with these processes indirectly or directly. Indirect process involves binding of nanomaterials to membrane and subsequent alteration in membrane integrity. Direct process involves redox-sensitive nanoparticle that binds to membrane-bound electron carriers and pulls out electrons from the transport chain. Fullerene and cerium dioxide nanoparticles are capable of interfering with the cellular energy transduction [205–207].

As mitochondria plays a role in regulating apoptosis (programmed cell death), nanomaterials are capable of impairing the regulatory of mitochondria. Quantum dots and silica nanoparticles induce developmental malformation in the zebra fish embryo heart. Gold nanoparticles induce malpigmentation in the eyes of zebrafish leading to altered swimming behaviour. Both these effects were reported to be due to interference with the apoptosis signalling [208–210].

Considering the toxicological effects of several nanoparticles, several *in vitro* and *in vivo* experimental studies have been done to identify the potential compounds which could combat the toxicity of specific nanoparticles. For example, *in vivo* studies have proved the efficacy of nanoparticles of bisdemethoxycurcumin analog in ameliorating MWCNT-induced toxicity [211]. It has been reported that fenugreek extract and quercetin combat petrol exhaust nanoparticle-induced lipid peroxidation and oxidative stress in red blood cells [212]. Cardioprotective effect of quercetin was reported in petrol exhaust nanoparticle-induced toxicity *in vitro* [213].

5.6. Phytotoxicity of Nanoparticles. Wang et al. have demonstrated the trans-generational effects of ceria nanoparticles using tomato plants. Tomato plants were treated with low concentrations of CeO_2 ($10 \text{ mg}\cdot\text{L}^{-1}$) and the seeds from the generation plant were collected and used to grow second-generation seedlings. The second-generation plants were relatively smaller and weaker, as indicated by their smaller biomass, lower water transpiration, and slightly higher reactive oxygen species content together with higher accumulation of ceria [214]. Similarly, ceria nanoparticles are toxic to wheat as indicated by morphological and biochemical effects, reductions in chlorophyll, delayed flowering, and increased catalase and superoxide dismutase activities [215]. Soilborne ceria nanoparticles show bioaccumulation in the roots and root nodules of soybeans. Ceria also shows bioaccumulation in the roots and shoots of corn plants [216, 217]. Ceria nanoparticles induces alteration in nutrient assimilation in corn and bioaccumulation in soybean [218, 219].

Titania, silver, and zinc oxide nanoparticles were added to soil samples. The amended soil was used as media for the legume *Medicago truncatula*. These nanoparticles were reported to reduce the root nodulation in legume and also altered the microbial community in the soil [218]. Soil under field conditions spiked with titania and zinc oxide nanoparticles bind into the cells of wheat plants. Titania nanoparticles bind strongly to the cell wall of wheat plant cells and decreased the biomass. Zinc oxide nanoparticles that are absorbed not only decreased the biomass but also decreased the activities of soil enzymes protease, catalase, and peroxidase showing the adverse effect on soil quality [219].

Engineered iron oxide nanoparticles were absorbed and accumulated by pumpkin (*Cucurbita maxima*) plants [220]. Arbuscular mycorrhizal fungi (AMF) establish a mutualistic symbiosis with majority of the terrestrial plants. Iron oxide nanoparticles significantly reduced mycorrhizal clover biomass by reducing the glomalin content and root nutrient acquisition of AMF [221]. In plants, nanoscale aluminium was reported to induce phytotoxicity by suppressing rhizogenesis probably through high solubility [222].

6. Nanomaterials as Vectors for Other Contaminants

Several contaminants such as radioactive elements, polychlorinated compounds [223], potentially toxic elements (PTE) [224], and pesticides [225] present in the sediments or suspended as solids in water undergo sorption on nanoparticle's surface through physical sorption (which involves van der Waals interaction), electrostatic interaction, ion-exchange, chemical adsorption (involves bond formation) etc. The sorption process of various contaminants may passivate the nanoparticle's active surfaces affecting environmental behaviour and toxicity of not only of the nanoparticle but also the adsorbed contaminants by influencing their dissolution and solubility kinetics [226]. Nanoparticles act as carriers of contaminants and enter in the organism as a particle-contaminant complex. Thus, the complexed contaminants are released inside the organisms, thereby increasing its bioavailability and toxicity by the "Trojan horse effect."

Recent studies have reported the following:

- (1) Adsorption of polybrominated diphenyl ethers on titanium dioxide and zinc oxide nanoparticles through aromatic ether groups and Br suppresses the agglomeration of nanoparticles in natural water resulting in enhanced transport and exposure of pollutants to aquatic organisms [227, 228].
- (2) Sorption of certain antibiotics like levofloxacin and ciprofloxacin on graphene oxide (GO) enhances the mobility and transport of these antibiotics through porous media which may increase their risks to ecological receptors and could also cause groundwater contamination [229].
- (3) The pluronic acid-modified single-walled carbon nanotubes (PA-SWCNTs) act as a carrier in the

porous media and facilitate the transport of gold nanoparticles [230].

- (4) Similarly, the adsorption of polycyclic aromatic hydrocarbons, PAHs (naphthalene, phenanthrene), and aldrin through electrostatic attraction and surface complexation process on TiO₂ and MWCNTs surface increases the aqueous phase concentration of these hydrophobic organic pollutants as compared to the “real” partitioning due to the octanol-water partitioning [231].
- (5) The hydrophobic dissolved organic matter (DOM) coating on Ag-Ag₂S and CeO₂ nanoparticles reduced the adsorption capacity of methylene blue by around 54% and 70%, respectively. The blockage of nanoparticle surface active sites with DOM reduces the adsorption of dye [232].
- (6) The adsorption of organochlorines (atrazine, hexachlorobenzene, pentachlorobenzene, and 3, 3', 4, 4'-tetrachlorobiphenyl) by TiO₂ nanoparticles increases the bioaccumulation of these contaminants to algae [223].
- (7) The co-exposure of nanoparticles (such as TiO₂, Ag, Al₂O₃, graphene, and CNTs) with PAHs, organochlorine pesticides (OCPs), and polybrominated diphenyl ether (PBDE) in soil to *Ipomoea* aquatic, *Cucumis sativus* L., *Zea mays* L., *Spinacia oleracea* L., and *Cucurbita moschata* promotes the uptake and accumulation of organic contaminants in crop plants [233].
- (8) Co-exposure of graphene nanoparticles at 50 mg/kg level (4–20 nm-size) substantially increased the bioaccumulation of organic contaminants through adsorption, followed by co-transfer into crop tissues [233].
- (9) Pulmonary surfactant (PS) altered the air-liquid interfacial properties of carbon nanoparticles which may increase the pulmonary risk of atmospheric exposure of both PAHs and carbon nanoparticles [234].
- (10) Co-exposure of ZnO Nanoparticles and Pb²⁺ enhanced the Pb accumulation in all major organs of mice, induced excessive production of hepatic ROS and pro-inflammatory cytokines, and aggravated liver injury [235].

Apart from the above, there is an emerging concern for microplastics in the aquatic ecosystems, which upon interaction with nanoparticles can either passivate or accelerate the environmental hazards. The interaction of microplastics with nanoparticles can modify the properties such as size, surface chemistry, stability, conductivity, solubility, persistence resulting fate, transport, and toxicity of both nanoparticles and microplastics [236]. Polystyrene (PS) microplastics can effectively remove the Ag nanoparticles in the aquatic environment by monolayer adsorption through electrostatic interaction [237]. The co-presence of TiO₂ nanoparticles in polystyrene microplastic

suspensions decreases microplastic transport and promotes their rapid deposition in quartz sand by the formation of heteroagglomerates [238]. It has been reported that the co-exposure of PS and Ag nanoparticles to *Chlamydomonas reinhardtii* and *Ochromonas danica* stimulated the toxicity of pollutants to freshwater algae through synergistic interactions [239]. Similarly, the chronic exposure of MP and Au nanoparticle mixture to planktonic crustacean *Daphnia magna* hinders development, reproduction, and eventually leads to death.

Thus, it could be concluded that the adsorption of contaminants on the surface of the nanoparticles not only modifies the environmental fate of both the contaminant and nanoparticles but also alters the ecotoxicity of nanoparticle-contaminant complexes.

7. Trophic Transfer of Nanoparticles

The biomagnification of gold nanomaterials in a simulated tobacco (*Nicotiana tabacum*)-tobacco hornworm (*Manduca sexta*) caterpillar food chain [240] has been reported. This biomagnification raises concerns regarding the potential for humans to be exposed to nanomaterials via trophic transfer.

This work was followed by other studies that reported trophic transfer of nanomaterials in other simulated food chains including:

- (1) Earthworm (*Eisenia fetida*) to Bullfrog (*Rana catesbeiana*) [241].
- (2) Zucchini (*Cucurbita pepo*) to Cricket (*Acheta domestica*) [242].
- (3) Lettuce (*Lactuca sativa*) to Cricket to Darkling beetles (*Tenebrionoidea*) [243].

In spite of the fact that each of the aforementioned studies reported trophic transfer, the concentration of nanomaterials transferred were small and the biomagnification reported in the tomato-hornworm study was not observed. However, recently, the biomagnification of nanomaterials was again reported in a terrestrial food chain, with CeO₂ nanomaterials accumulated in kidney bean (*Phaseolus vulgaris*), plants biomagnifying in Mexican bean beetles (*Epilachnavarivestis*), as well as in consumers of the beetles, spined soldier bugs (*Podisus maculiventris*) [244]. Altogether, these studies indicate that trophic transfer of nanomaterials occur at least in some small amount in most terrestrial food chains, with certain terrestrial organisms and food chains being particularly susceptible to biomagnification, although the reasons that some organisms are more likely to biomagnify nanomaterials remain unclear.

8. Conclusion

There are only limited knowledge and limited review papers focusing on the environmental hazards and exposure of nanomaterials, which in turn creates large uncertainties in understanding the risk. This type of review will result in the

database showing the risk of eco-nanotoxicity, the opportunities and challenges in the determination and the prevention of eco-nanotoxicity. The degrees of uncertainties in the data can be analysed by using this type of review. Regulatory agencies can avail this type of review to rightly classify the nanoparticles and to manage nanomaterials as compared to their bulk counterparts. Hence, our review systematically focused on the source of exposure, environmental and biological dynamics, and the adverse effects of nanoparticles whose production rate is constantly in hike. This approach will reveal many research targets for nanotechnology community and will be a frontline to address environmental issues. This review will afford a reference for future studies and guide researchers for structuring objectives to tackle the nanotoxicology-related issues and to reduce further emissions. We infer that this review will direct the scientists and researchers towards a new era of green nanotechnology.

9. Future Perspectives

Suitable analytical metrology for detecting the physicochemical properties and the quanta of transformed nanoparticles in the environmental matrices is needed. This can to a certain extent be achieved by integrating and customising the currently available sophisticated techniques. Also, suitable methodology to isolate and quantify the bioavailable proportions of nanoparticles is essential for determining the toxicity in aquatic and terrestrial organisms. As the marketing of nanoparticle-based products are in hike, strict testing and regulation should be enforced to regulate the production, handling, and disposal of nanoparticles. As the research on nanobiotechnology is increasing, the animal experimentation pertaining to biomedical applications of nanomaterials should be properly regulated. More focused research and experiments can be done on the teratogenic effects of different types of nanoparticles. Environmental-friendly protocols can be followed during the synthesis, applications, and the disposal of wastewater and devices fabricated with nanoparticles.

Data Availability

The datasets used and/or investigated during the current study are available from the corresponding authors upon reasonable request.

Conflicts of Interest

The authors declare that they have no conflicts of interest.

Acknowledgments

Iffath B gratefully acknowledges the AICTE (All India Council for Technical Education) for providing the AICTE Doctoral Fellowship (ADF)-student ID S2022311. The authors spent their own fund to write this review.

References

- [1] M. J. Hajipour, K. M. Fromm, A. Akbar Ashkarran et al., "Antibacterial properties of nanoparticles," *Trends in Biotechnology*, vol. 30, no. 10, pp. 499–511, 2012.
- [2] J. Shi, P. W. Kantoff, R. Wooster, and O. C. Farokhzad, "Cancer nanomedicine: progress, challenges and opportunities," *Nature Reviews Cancer*, vol. 17, no. 1, pp. 20–37, 2017.
- [3] V. Weissig, T. K. Pettinger, and N. Murdock, "Nanopharmaceuticals (part 1): products on the market," *International Journal of Nanomedicine*, vol. 9, pp. 4357–4373, 2014.
- [4] M. Colombo, S. Carregal-Romero, M. F. Casula et al., "Biological applications of magnetic nanoparticles," *Chemical Society Reviews*, vol. 41, no. 11, p. 4306, 2012.
- [5] E. C. Dreaden, A. M. Alkilany, X. Huang, C. J. Murphy, and M. A. El-Sayed, "The golden age: gold nanoparticles for biomedicine," *Chemical Society Reviews*, vol. 41, no. 7, pp. 2740–2779, 2012.
- [6] J. W. Rhim, H. M. Park, and C. S. Ha, "Bio-nanocomposites for food packaging applications," *Progress in Polymer Science*, vol. 38, no. 10–11, pp. 1629–1652, 2013.
- [7] N. Sundaria, M. Singh, P. Upreti, R. P. Chauhan, J. P. Jaiswal, and A. Kumar, "Seed priming with iron oxide nanoparticles triggers iron acquisition and biofortification in wheat (*Triticum aestivum* L.) grains," *Journal of Plant Growth Regulation*, vol. 38, no. 1, pp. 122–131, 2019.
- [8] R. Grillo, L. F. Fraceto, M. J. Amorim, J. J. Scott-Fordsmand, R. Schoonjans, and Q. Chaudhry, "Ecotoxicological and regulatory aspects of environmental sustainability of nanopesticides," *Journal of Hazardous Materials*, vol. 404, Article ID 124148, 2021.
- [9] P. J. Lu, S. W. Fang, W. L. Cheng, S. C. Huang, M. C. Huang, and H. F. Cheng, "Characterization of titanium dioxide and zinc oxide nanoparticles in sunscreen powder by comparing different measurement methods," *Journal of Food and Drug Analysis*, vol. 26, no. 3, pp. 1192–1200, 2018.
- [10] S. J. Klaine, A. A. Koelmans, N. Horne et al., "Paradigms to assess the environmental impact of manufactured nanomaterials," *Environmental Toxicology and Chemistry*, vol. 31, no. 1, pp. 3–14, 2012.
- [11] I. K. Januariyasa, I. D. Ana, and Y. Yusuf, "Nanofibrous poly (vinyl alcohol)/chitosan contained carbonated hydroxyapatite nanoparticles scaffold for bone tissue engineering," *Materials Science and Engineering: C*, vol. 107, Article ID 110347, 2020.
- [12] F. J. Heiligtag and M. Niederberger, "The fascinating world of nanoparticle research," *Materials Today*, vol. 16, no. 7–8, pp. 262–271, 2013.
- [13] A. Ito, M. Shinkai, H. Honda, and T. Kobayashi, "Medical application of functionalized magnetic nanoparticles," *Journal of Bioscience and Bioengineering*, vol. 100, no. 1, pp. 1–11, 2005.
- [14] E. Semenzin, E. Lanzellotto, D. Hristozov et al., "Species sensitivity weighted distribution for ecological risk assessment of engineered nanomaterials: the n-TiO₂ case study," *Environmental Toxicology and Chemistry*, vol. 34, no. 11, pp. 2644–2659, 2015.
- [15] S. Foss Hansen, L. R. Heggelund, P. Revilla Besora, A. Mackevica, A. Boldrin, and A. Baun, "Nanoproducts-what is actually available to European consumers?" *Environmental Sciences: Nano*, vol. 3, no. 1, pp. 169–180, 2016.

- [16] M. Bundschuh, J. Filser, S. Luderwald et al., "Nanoparticles in the environment: where do we come from, where do we go to?" *Environmental Sciences Europe*, vol. 30, no. 1, p. 6, 2018.
- [17] B. Nowack and T. D. Bucheli, "Occurrence, behavior and effects of nanoparticles in the environment," *Environmental Pollution*, vol. 150, no. 1, pp. 5–22, 2007.
- [18] A. Freixa, V. Acuña, J. Sanchis, M. Farre, D. Barcelo, and S. Sabater, "Ecotoxicological effects of carbon-based nanomaterials in aquatic organisms," *Science of the Total Environment*, vol. 619–620, pp. 328–337, 2018.
- [19] E. J. Petersen, L. Zhang, N. T. Mattison et al., "Potential release pathways, environmental fate, and ecological risks of carbon nanotubes," *Environmental Science & Technology*, vol. 45, no. 23, pp. 9837–9856, 2011.
- [20] M. Baalousha, Y. Ju-Nam, P. A. Cole et al., "Characterization of cerium oxide nanoparticles—part 1: size measurements," *Environmental Toxicology and Chemistry*, vol. 31, no. 5, pp. 983–993, 2012.
- [21] G. V. Lowry, K. B. Gregory, S. C. Apte, and J. R. Lead, "Transformations of nanomaterials in the environment," *Environmental Science & Technology*, vol. 46, no. 13, pp. 6893–6899, 2012.
- [22] H. Selck, R. D. Handy, T. F. Fernandes, S. J. Klaine, and E. J. Petersen, "Nanomaterials in the aquatic environment: a European Union–United States perspective on the status of ecotoxicity testing, research priorities, and challenges ahead," *Environmental Toxicology and Chemistry*, vol. 35, no. 5, pp. 1055–1067, 2016.
- [23] F. Gottschalk, T. Sonderer, R. W. Scholz, and B. Nowack, "Modeled environmental concentrations of engineered nanomaterials (TiO₂, ZnO, Ag, CNT, fullerenes) for different regions," *Environmental Science & Technology*, vol. 43, no. 24, pp. 9216–9222, 2009.
- [24] F. Gottschalk, T. Sun, and B. Nowack, "Environmental concentrations of engineered nanomaterials: review of modeling and analytical studies," *Environmental Pollution*, vol. 181, pp. 287–300, 2013.
- [25] T. Tolaymat, A. El Badawy, A. Genaidy, W. Abdelraheem, and R. Sequeira, "Analysis of metallic and metal oxide nanomaterial environmental emissions," *Journal of Cleaner Production*, vol. 143, pp. 401–412, 2017.
- [26] F. Gottschalk and B. Nowack, "The release of engineered nanomaterials to the environment," *Journal of Environmental Monitoring*, vol. 13, no. 5, p. 1145, 2011.
- [27] A. A. Keller, S. McFerran, A. Lazareva, and S. Suh, "Global life cycle releases of engineered nanomaterials," *Journal of Nanoparticle Research*, vol. 15, no. 6, p. 1692, 2013.
- [28] T. Y. Sun, N. A. Bornhoft, K. Hungerbuhler, and B. Nowack, "Dynamic probabilistic modeling of environmental emissions of engineered nanomaterials," *Environmental Science & Technology*, vol. 50, no. 9, pp. 4701–4711, 2016.
- [29] N. C. Mueller and B. Nowack, "Exposure modeling of engineered nanoparticles in the environment," *Environmental Science & Technology*, vol. 42, no. 12, pp. 4447–4453, 2008.
- [30] A. G. Schultz, D. Boyle, D. Chamot et al., "Aquatic toxicity of manufactured nanomaterials: challenges and recommendations for future toxicity testing," *Environmental Chemistry*, vol. 11, no. 3, p. 207, 2014.
- [31] F. Arul Prakash, G. J. Dushendra Babu, M. Lavanya, K. Shenbaga Vidhya, and T. Devasena, "Toxicity studies of aluminium oxide nanoparticles in cell lines," *International Journal of Nanotechnology and Applications*, vol. 5, no. 2, pp. 99–107, 2011.
- [32] A. P. Francis and T. Devasena, "Toxicity of carbon nanotubes: a review," *Toxicology and Industrial Health*, vol. 34, no. 3, pp. 200–210, 2018.
- [33] A. P. Francis, S. Ganapathy, V. R. Palla, P. B. Murthy, S. Ramaprabhu, and T. Devasena, "One time nose-only inhalation of MWCNTs: exploring the mechanism of toxicity by intermittent sacrifice in Wistar rats," *Toxicology Reports*, vol. 2, pp. 111–120, 2015.
- [34] P. J. Borm, "Toxicity and occupational health hazards of coal fly ash (CFA). A review of data and comparison to coal mine dust," *Annals of Occupational Hygiene*, vol. 41, no. 6, pp. 659–676, 1997.
- [35] B. Sambandam, E. Palanisami, R. Abbugounder, B. Prakhya, and D. Thiyagarajan, "Characterizations of coal fly ash nanoparticles and induced in vitro toxicity in cell lines," *Journal of Nanoparticle Research*, vol. 16, no. 2, p. 2217, 2014.
- [36] D. Mohan, T. Devasena, and P. B. Murthy, "Toxicity of exhaust nanoparticles," *African Journal of Pharmacy and Pharmacology*, vol. 7, no. 7, pp. 318–331, 2013.
- [37] M. Durga, F. Arul Prakash, A. Rajasekar, and T. Devasena, "Potential cell line toxicity of environmental nanoparticles," *International Research Journal of Pharmacy*, vol. 3, no. 1, pp. 145–149, 2012.
- [38] M. Durga, S. Nathiya, A. Rajasekar, and T. Devasena, "Effects of ultrafine petrol exhaust particles on cytotoxicity, oxidative stress generation, DNA damage and inflammation in human A549 lung cells and murine RAW 264.7 macrophages," *Environmental Toxicology and Pharmacology*, vol. 38, no. 2, pp. 518–530, 2014.
- [39] S. Kashiwada, "Distribution of nanoparticles in the see-through medaka (*Oryzias latipes*)," *Environmental Health Perspectives*, vol. 114, no. 11, pp. 1697–1702, 2006.
- [40] D. W. Grainger and D. G. Castner, "Nanobiomaterials and nanoanalysis: opportunities for improving the science to benefit biomedical technologies," *Advanced Materials*, vol. 20, no. 5, pp. 867–877, 2008.
- [41] A. B. Boxall, K. Tiede, and Q. Chaudhry, "Engineered nanomaterials in soils and water: how do they behave and could they pose a risk to human health?" *Nanomedicine*, vol. 2, no. 6, pp. 919–927, 2007.
- [42] A. Thit, G. T. Banta, and H. Selck, "Bioaccumulation, sub-cellular distribution and toxicity of sediment-associated copper in the ragworm *Nereis diversicolor*: the relative importance of aqueous copper, copper oxide nanoparticles and microparticles," *Environmental Pollution*, vol. 202, pp. 50–57, 2015.
- [43] T. Y. Sun, F. Gottschalk, K. Hungerbuhler, and B. Nowack, "Comprehensive probabilistic modelling of environmental emissions of engineered nanomaterials," *Environmental Pollution*, vol. 185, pp. 69–76, 2014.
- [44] G. Sun, Y. Zhang, L. Pei et al., "Chemometric QSAR modeling of acute oral toxicity of Polycyclic Aromatic Hydrocarbons (PAHs) to rat using simple 2D descriptors and interspecies toxicity modeling with mouse," *Ecotoxicology and Environmental Safety*, vol. 222, Article ID 112525, 2021.
- [45] F. Laborda, E. Bolea, G. Cepria et al., "Detection, characterization and quantification of inorganic engineered nanomaterials: a review of techniques and methodological approaches for the analysis of complex samples," *Analytica Chimica Acta*, vol. 904, pp. 10–32, 2016.
- [46] O. Borovinskaya, S. Gschwind, B. Hattendorf, M. Tanner, and D. Gunther, "Simultaneous mass quantification of nanoparticles of different composition in a mixture by

- microdroplet generator-ICPTOFMS,” *Analytical Chemistry*, vol. 86, no. 16, pp. 8142–8148, 2014.
- [47] A. Philippe and G. E. Schaumann, “Evaluation of hydrodynamic chromatography coupled with UV-visible, fluorescence and inductively coupled plasma mass spectrometry detectors for sizing and quantifying colloids in environmental media,” *PLoS One*, vol. 9, no. 2, Article ID e90559, 2014.
- [48] B. Meisterjahn, S. Wagner, F. Von der Kammer, D. Hennecke, and T. Hofmann, “Silver and gold nanoparticle separation using asymmetrical flow-field flow fractionation: influence of run conditions and of particle and membrane charges,” *Journal of Chromatography A*, vol. 1440, pp. 150–159, 2016.
- [49] R. B. Reed, C. P. Higgins, P. Westerhoff, S. Tadjiki, and J. F. Ranville, “Overcoming challenges in analysis of poly-disperse metal-containing nanoparticles by single particle inductively coupled plasma mass spectrometry,” *Journal of Analytical Atomic Spectrometry*, vol. 27, no. 7, p. 1093, 2012.
- [50] M. Tharaud, A. P. Gondikas, M. F. Benedetti, F. Von der Kammer, T. Hofmann, and G. Cornelis, “TiO₂ nanomaterial detection in calcium rich matrices by spICPMS. A matter of resolution and treatment,” *Journal of Analytical Atomic Spectrometry*, vol. 32, no. 7, pp. 1400–1411, 2017.
- [51] R. C. Merrifield, K. P. Arkill, R. E. Palmer, and J. R. Lead, “A high resolution study of dynamic changes of Ce₂O₃ and CeO₂ nanoparticles in complex environmental media,” *Environmental Science & Technology*, vol. 51, no. 14, pp. 8010–8016, 2017.
- [52] R. C. Merrifield, C. Stephan, and J. R. Lead, “Quantification of Au nanoparticle bio uptake and distribution to freshwater algae using single cell-ICP-MS,” *Environmental Science & Technology*, vol. 52, no. 4, pp. 2271–2277, 2018.
- [53] E. P. Gray, J. G. Coleman, A. J. Bednar, A. J. Kennedy, J. F. Ranville, and C. P. Higgins, “Extraction and analysis of silver and gold nanoparticles from biological tissues using single particle inductively coupled plasma mass spectrometry,” *Environmental Science & Technology*, vol. 47, no. 24, pp. 14315–14323, 2013.
- [54] B. D. Johnston, T. M. Scown, J. Moger et al., “Bioavailability of nanoscale metal oxides TiO₂, CeO₂, and ZnO to fish,” *Environmental Science & Technology*, vol. 44, no. 3, pp. 1144–1151, 2010.
- [55] T. Cresswell, M. Metian, L. A. Golding, and M. D. Wood, “Aquatic live animal radiotracing studies for ecotoxicological applications: addressing fundamental methodological deficiencies,” *Journal of Environmental Radioactivity*, vol. 178–179, pp. 453–460, 2017.
- [56] M. S. Hull, P. Chaurand, J. Rose et al., “Filter-feeding bivalves store and biodeposit colloiddally stable gold nanoparticles,” *Environmental Science & Technology*, vol. 45, no. 15, pp. 6592–6599, 2011.
- [57] A. P. Gondikas, F. V. D. Kammer, R. B. Reed, S. Wagner, J. F. Ranville, and T. Hofmann, “Release of TiO₂ nanoparticles from sunscreens into surface waters: a one-year survey at the old Danube recreational Lake,” *Environmental Science & Technology*, vol. 48, no. 10, pp. 5415–5422, 2014.
- [58] J. Garcia-Alonso, F. R. Khan, S. K. Misra et al., “Cellular internalization of silver nanoparticles in gut epithelia of the estuarine polychaete *Nereis diversicolor*,” *Environmental Science & Technology*, vol. 45, no. 10, pp. 4630–4636, 2011.
- [59] X. Zhu, L. Zhu, Y. Chen, and S. Tian, “Acute toxicities of six manufactured nanomaterial suspensions to *Daphnia magna*,” *Journal of Nanoparticle Research*, vol. 11, no. 1, pp. 67–75, 2009.
- [60] M. Heinlaan, A. Kahru, K. Kasemets, B. Arbeille, G. Prensier, and H. C. Dubourguier, “Changes in the *Daphnia magna* midgut upon ingestion of copper oxide nanoparticles: a transmission electron microscopy study,” *Water Research*, vol. 45, no. 1, pp. 179–190, 2011.
- [61] R. D. Handy, G. Cornelis, T. Fernandes et al., “Ecotoxicity test methods for engineered nanomaterials: practical experiences and recommendations from the bench,” *Environmental Toxicology and Chemistry*, vol. 31, no. 1, pp. 15–31, 2012.
- [62] R. C. Merrifield, C. Stephan, and J. R. Lead, “Single-particle inductively coupled plasma mass spectroscopy analysis of size and number concentration in mixtures of monometallic and bimetallic (core-shell) nanoparticles,” *Talanta*, vol. 162, pp. 130–134, 2017.
- [63] M. N. Croteau, S. K. Misra, S. N. Luoma, and E. Valsami-Jones, “Bioaccumulation and toxicity of CuO nanoparticles by a freshwater invertebrate after waterborne and dietborne exposures,” *Environmental Science & Technology*, vol. 48, no. 18, pp. 10929–10937, 2014.
- [64] A. Boxall, Q. Chaudhry, C. Sinclair et al., “Current and future predicted environmental exposure to engineered nanoparticles,” *Safety of Nanomaterials Interdisciplinary Research Centre Report*, vol. 196111, pp. 1–13, 2007.
- [65] B. Park, K. Donaldson, R. Duffin et al., “Hazard and risk assessment of a nanoparticulate cerium oxide-based diesel fuel additive—a case study,” *Inhalation Toxicology*, vol. 20, no. 6, pp. 547–566, 2008.
- [66] S. A. Blaser, M. Scheringer, M. MacLeod, and K. Hungerbühler, “Estimation of cumulative aquatic exposure and risk due to silver: contribution of nano-functionalized plastics and textiles,” *Science of the Total Environment*, vol. 390, no. 2–3, pp. 396–409, 2008.
- [67] A. A. Koelmans, B. Nowack, and M. R. Wiesner, “Comparison of manufactured and black carbon nanoparticle concentrations in aquatic sediments,” *Environmental Pollution*, vol. 157, no. 4, pp. 1110–1116, 2009.
- [68] F. Gottschalk, T. Sonderer, R. W. Scholz, and B. Nowack, “Possibilities and limitations of modeling environmental exposure to engineered nanomaterials by probabilistic material flow analysis,” *Environmental Toxicology and Chemistry*, vol. 29, no. 5, pp. 1036–1048, 2010.
- [69] F. Gottschalk, R. W. Scholz, and B. Nowack, “Probabilistic material flow modeling for assessing the environmental exposure to compounds: methodology and an application to engineered nano-TiO₂ particles,” *Environmental Modelling & Software*, vol. 25, no. 3, pp. 320–332, 2010.
- [70] N. Musee, “Simulated environmental risk estimation of engineered nanomaterials: a case of cosmetics in Johannesburg City,” *Human & Experimental Toxicology*, vol. 30, no. 9, pp. 1181–1195, 2010.
- [71] A. C. Johnson, M. J. Bowes, A. Crossley et al., “An assessment of the fate, behaviour and environmental risk associated with sunscreen TiO₂ nanoparticles in UK field scenarios,” *Science of the Total Environment*, vol. 409, no. 13, pp. 2503–2510, 2011.
- [72] R. Arvidsson, S. Molander, and B. A. Sandén, “Particle flow analysis: exploring potential use phase emissions of titanium dioxide nanoparticles from sunscreen, paint, and cement,” *Journal of Industrial Ecology*, vol. 16, no. 3, pp. 343–351, 2012.

- [73] S. N. Luoma, F. R. Khan, and M. N. Croteau, "Bioavailability and bioaccumulation of metal-based engineered nanomaterials in aquatic environments: concepts and processes," *Frontiers of Nanoscience*, vol. 7, pp. 157–193, 2014.
- [74] J. Fabrega, R. Zhang, J. C. Renshaw, W. T. Liu, and J. R. Lead, "Impact of silver nanoparticles on natural marine biofilm bacteria," *Chemosphere*, vol. 85, no. 6, pp. 961–966, 2011.
- [75] N. S. Taylor, R. Merrifield, T. D. Williams, J. K. Chipman, J. R. Lead, and M. R. Viant, "Molecular toxicity of cerium oxide nanoparticles to the freshwater alga *Chlamydomonas reinhardtii* is associated with supra-environmental exposure concentrations," *Nanotoxicology*, vol. 10, no. 1, pp. 32–41, 2016.
- [76] T. Y. Sun, D. M. Mitrano, N. A. Bornhoft, M. Scheringer, K. Hungerbühler, and B. Nowack, "Envisioning nano release dynamics in a changing world: using dynamic probabilistic modeling to assess future environmental emissions of engineered nanomaterials," *Environmental Science & Technology*, vol. 51, no. 5, pp. 2854–2863, 2017.
- [77] A. M. Derfus, W. C. W. Chan, and S. N. Bhatia, "Probing the cytotoxicity of semiconductor quantum dots," *Nano Letters*, vol. 4, no. 1, pp. 11–18, 2004.
- [78] W. Lesniak, A. U. Bielinska, K. Sun et al., "Silver/dendrimer nanocomposites as biomarkers: fabrication, characterization, *in vitro* toxicity, and intracellular detection," *Nano Letters*, vol. 5, no. 11, pp. 2123–2130, 2005.
- [79] R. Liu, H. Y. Zhang, Z. X. Ji et al., "Development of structure-activity relationship for metal oxide nanoparticles," *Nanoscale*, vol. 5, no. 12, p. 5644, 2013.
- [80] L. Liu, M. Sun, H. Zhang et al., "Facet energy and reactivity versus cytotoxicity: the surprising behavior of CdS nanorods," *Nano Letters*, vol. 16, no. 1, pp. 688–694, 2016.
- [81] C. M. Sayes, "The relationships among structure, activity, and toxicity of engineered nanoparticles," *KONA Powder and Particle Journal*, vol. 31, no. 0, pp. 10–21, 2014.
- [82] Q. Abbas, B. Yousaf, M. U. Ali et al., "Transformation pathways and fate of engineered nanoparticles (ENPs) in distinct interactive environmental compartments: a review," *Environment International*, vol. 138, Article ID 105646, 2020.
- [83] J. R. Lead, G. E. Batley, P. J. J. Alvarez et al., "Nanomaterials in the environment: behavior, fate, bioavailability, and effects—an updated review," *Environmental Toxicology and Chemistry*, vol. 37, no. 8, pp. 2029–2063, 2018.
- [84] R. D. Handy, F. von der Kammer, J. R. Lead, M. Hasselov, R. Owen, and M. Crane, "The ecotoxicology and chemistry of manufactured nanoparticles," *Ecotoxicology*, vol. 17, no. 4, pp. 287–314, 2008.
- [85] S. W. Bian, I. A. Mudunkotuwa, T. Rupasinghe, and V. H. Grassian, "Aggregation and dissolution of 4 nm ZnO nanoparticles in aqueous environments: influence of pH, ionic strength, size, and adsorption of humic acid," *Langmuir*, vol. 27, no. 10, pp. 6059–6068, 2011.
- [86] M. Baalousha, G. Cornelis, T. A. J. Kuhlbusch et al., "Modeling nanomaterial fate and uptake in the environment: current knowledge and future trends," *Environmental Sciences: Nano*, vol. 3, no. 2, pp. 323–345, 2016.
- [87] X. Yang, Q. Wang, X. Qu, and W. Jiang, "Bound and unbound humic acids perform different roles in the aggregation and deposition of multi-walled carbon nanotubes," *Science of the Total Environment*, vol. 586, pp. 738–745, 2017.
- [88] S. Diegoli, A. L. Manciuola, S. Begum, I. P. Jones, J. R. Lead, and J. A. Preece, "Interaction between manufactured gold nanoparticles and naturally occurring organic macromolecules," *Science of the Total Environment*, vol. 402, no. 1, pp. 51–61, 2008.
- [89] R. C. Merrifield, C. Stephan, and J. Lead, "Determining the concentration dependent transformations of Ag nanoparticles in complex media: using SP-ICP-MS and Au@Ag core-shell nanoparticles as tracers," *Environmental Science & Technology*, vol. 51, no. 6, pp. 3206–3213, 2017.
- [90] J. E. Ward and D. J. Kach, "Marine aggregates facilitate ingestion of nanoparticles by suspension-feeding bivalves," *Marine Environmental Research*, vol. 68, no. 3, pp. 137–142, 2009.
- [91] A. Alabresm, S. Mirshahghassemi, G. T. Chandler, A. W. Decho, and J. Lead, "Use of PVP-coated magnetite nanoparticles to ameliorate oil toxicity to an estuarine meiobenthic copepod and stimulate the growth of oil-degrading bacteria," *Environmental Sciences: Nano*, vol. 4, no. 9, pp. 1859–1865, 2017.
- [92] A. M. E. Badawy, T. P. Luxton, R. G. Silva, K. G. Scheckel, M. T. Suidan, and T. M. Tolaymat, "Impact of environmental conditions (pH, ionic strength, and electrolyte type) on the surface charge and aggregation of silver nanoparticles suspensions," *Environmental Science & Technology*, vol. 44, no. 4, pp. 1260–1266, 2010.
- [93] R. D. Handy and F. B. Eddy, "Transport of solutes across biological membranes in eukaryotes: an environmental perspective," *IUPAC series on analytical and physical chemistry of environmental systems*, vol. 9, pp. 337–356, 2004.
- [94] J. T. Quik, M. C. Stuart, M. Wouterse, W. Peijnenburg, A. J. Hendriks, and D. van de Meent, "Natural colloids are the dominant factor in the sedimentation of nanoparticles," *Environmental Toxicology and Chemistry*, vol. 31, no. 5, pp. 1019–1022, 2012.
- [95] J. T. Quik, J. A. Vonk, S. F. Hansen, A. Baun, and D. Van De Meent, "How to assess exposure of aquatic organisms to manufactured nanoparticles?" *Environment International*, vol. 37, no. 6, pp. 1068–1077, 2011.
- [96] Z. M. Xiu, J. Ma, and P. J. J. Alvarez, "Differential effect of common ligands and molecular oxygen on antimicrobial activity of silver nanoparticles versus silver ions," *Environmental Science & Technology*, vol. 45, no. 20, pp. 9003–9008, 2011.
- [97] N. M. Franklin, N. J. Rogers, S. C. Apte, G. E. Batley, G. E. Gadd, and P. S. Casey, "Comparative toxicity of nanoparticulate ZnO, bulk ZnO, and ZnCl₂ to a freshwater microalga (*Pseudokirchneriella subcapitata*): the importance of particle solubility," *Environmental Science & Technology*, vol. 41, no. 24, pp. 8484–8490, 2007.
- [98] N. J. Rogers, N. M. Franklin, S. C. Apte et al., "Physico-chemical behaviour and algal toxicity of nanoparticulate CeO₂ in freshwater," *Environmental Chemistry*, vol. 7, no. 1, p. 50, 2010.
- [99] A. Tsiola, P. Pitta, A. J. Callol et al., "The impact of silver nanoparticles on marine plankton dynamics: dependence on coating, size and concentration," *Science of the Total Environment*, vol. 601–602, pp. 1838–1848, 2017.
- [100] C. Toncelli, K. Mylona, I. Kalantzi et al., "Silver nanoparticles in seawater: a dynamic mass balance at part per trillion silver concentrations," *Science of the Total Environment*, vol. 601–602, pp. 15–21, 2017.
- [101] A. S. Adeleye, S. Pokhrel, L. Madler, and A. A. Keller, "Influence of nanoparticle doping on the colloidal stability and toxicity of copper oxide nanoparticles in synthetic and natural waters," *Water Research*, vol. 132, pp. 12–22, 2018.

- [102] Y. Xiao, M. G. Vijver, and W. J. Peijnenburg, "Impact of water chemistry on the behavior and fate of copper nanoparticles," *Environmental Pollution*, vol. 234, pp. 684–691, 2018.
- [103] S. Leclerc and K. J. Wilkinson, "Bioaccumulation of nanosilver by *Chlamydomonas reinhardtii*-nanoparticle or the free ion?" *Environmental Science & Technology*, vol. 48, no. 1, pp. 358–364, 2014.
- [104] C. Levard, E. M. Hotze, B. P. Colman et al., "Sulfidation of silver nanoparticles: natural antidote to their toxicity," *Environmental Science & Technology*, vol. 47, no. 23, pp. 13440–13448, 2013.
- [105] B. Kim, C. S. Park, M. Murayama, and M. F. Hochella, "Discovery and characterization of silver sulfide nanoparticles in final sewage sludge products," *Environmental Science & Technology*, vol. 44, no. 19, pp. 7509–7514, 2010.
- [106] R. Kaegi, A. Voegelin, B. Sinnert et al., "Behavior of metallic silver nanoparticles in a pilot wastewater treatment plant," *Environmental Science & Technology*, vol. 45, no. 9, pp. 3902–3908, 2011.
- [107] G. P. Devi, K. B. A. Ahmed, M. S. Varsha et al., "Sulfidation of silver nanoparticle reduces its toxicity in zebrafish," *Aquatic Toxicology*, vol. 158, pp. 149–156, 2015.
- [108] R. Ma, C. Levard, J. D. Judy et al., "Fate of zinc oxide and silver nanoparticles in a pilot wastewater treatment plant and in processed biosolids," *Environmental Science & Technology*, vol. 48, no. 1, pp. 104–112, 2014.
- [109] R. Ma, J. Stegemeier, C. Levard et al., "Sulfidation of copper oxide nanoparticles and properties of resulting copper sulfide," *Environmental Sciences: Nano*, vol. 1, no. 4, pp. 347–357, 2014.
- [110] X. Li, K. Schirmer, L. Bernard, L. Sigg, S. Pillai, and R. Behra, "Silver nanoparticle toxicity and association with the alga *Euglena gracilis*," *Environmental Sciences: Nano*, vol. 2, no. 6, pp. 594–602, 2015.
- [111] N. Grillet, D. Manchon, E. Cottancin et al., "Photo-oxidation of individual silver nanoparticles: a real-time tracking of optical and morphological changes," *Journal of Physical Chemistry C*, vol. 117, no. 5, pp. 2274–2282, 2013.
- [112] I. Romer, Z. W. Wang, R. C. Merrifield, R. E. Palmer, and J. Lead, "High resolution STEM-EELS study of silver nanoparticles exposed to light and humic substances," *Environmental Science & Technology*, vol. 50, no. 5, pp. 2183–2190, 2016.
- [113] G. Cornelis, K. Hund-Rinke, T. Kuhlbusch, N. Van den Brink, and C. Nickel, "Fate and bioavailability of engineered nanoparticles in soils: a review," *Critical Reviews in Environmental Science and Technology*, vol. 44, no. 24, pp. 2720–2764, 2014.
- [114] H. F. Lecoanet, J. Y. Bottero, and M. R. Wiesner, "Laboratory assessment of the mobility of nanomaterials in porous media," *Environmental Science & Technology*, vol. 38, no. 19, pp. 5164–5169, 2004.
- [115] D. P. Jaisi, N. B. Saleh, R. E. Blake, and M. Elimelech, "Transport of single-walled carbon nanotubes in porous media: filtration mechanisms and reversibility," *Environmental Science & Technology*, vol. 42, no. 22, pp. 8317–8323, 2008.
- [116] J. Fang, X. Q. Shan, B. Wen, J. M. Lin, and G. Owens, "Stability of titania nanoparticles in soil suspensions and transport in saturated homogeneous soil columns," *Environmental Pollution*, vol. 157, no. 4, pp. 1101–1109, 2009.
- [117] L. R. Heggelund, M. Diez-Ortiz, S. Lofst et al., "Soil pH effects on the comparative toxicity of dissolved zinc, nano and nano ZnO to the earthworm *Eisenia fetida*," *Nanotoxicology*, vol. 8, no. 5, pp. 559–572, 2014.
- [118] E. Lombi, E. Donner, E. Tavakkoli et al., "Fate of zinc oxide nanoparticles during anaerobic digestion of wastewater and post-treatment processing of sewage sludge," *Environmental Science & Technology*, vol. 46, no. 16, pp. 9089–9096, 2012.
- [119] P. Wang, N. W. Menzies, E. Lombi et al., "Fate of ZnO nanoparticles in soils and cowpea (*Vigna unguiculata*)," *Environmental Science & Technology*, vol. 47, no. 23, pp. 13822–13830, 2013.
- [120] D. A. Navarro, R. S. Kookana, M. J. McLaughlin, and J. K. Kirby, "Fullerol as a potential pathway for mineralization of fullerene nanoparticles in biosolid-amended soils," *Environmental Science and Technology Letters*, vol. 3, no. 1, pp. 7–12, 2016.
- [121] R. S. Chouhan, A. Qureshi, B. Yagci, M. A. Gulgun, V. Ozguz, and J. H. Niazi, "Biotransformation of multi-walled carbon nanotubes mediated by nanomaterial resistant soil bacteria," *Chemical Engineering Journal*, vol. 298, pp. 1–9, 2016.
- [122] A. Praetorius, J. Labille, M. Scheringer, A. Thill, K. Hungerbuhler, and J. Y. Bottero, "Heteroaggregation of titanium dioxide nanoparticles with model natural colloids under environmentally relevant conditions," *Environmental Science & Technology*, vol. 48, no. 18, pp. 10690–10698, 2014.
- [123] G. Cornelis, B. Ryan, M. J. McLaughlin, J. K. Kirby, D. Beak, and D. Chittleborough, "Solubility and batch retention of CeO₂ nanoparticles in soils," *Environmental Science & Technology*, vol. 45, no. 7, pp. 2777–2782, 2011.
- [124] G. Cornelis, C. DooletteMadeleine Thomas, M. J. McLaughlin, J. K. Kirby, D. G. Beak, and D. Chittleborough, "Retention and dissolution of engineered silver nanoparticles in natural soils," *Soil Science Society of America Journal*, vol. 76, no. 3, pp. 891–902, 2012.
- [125] B. M. Smith, D. J. Pike, M. O. Kelly, and J. A. Nason, "Quantification of heteroaggregation between citrate-stabilized gold nanoparticles and hematite colloids," *Environmental Science & Technology*, vol. 49, no. 21, pp. 12789–12797, 2015.
- [126] K. A. Huynh, J. M. McCaffery, and K. L. Chen, "Heteroaggregation of multiwalled carbon nanotubes and hematite nanoparticles: rates and mechanisms," *Environmental Science & Technology*, vol. 46, no. 11, pp. 5912–5920, 2012.
- [127] J. Labille, C. Harns, J. Y. Bottero, and J. Brant, "Heteroaggregation of titanium dioxide nanoparticles with natural clay colloids," *Environmental Science & Technology*, vol. 49, no. 11, pp. 6608–6616, 2015.
- [128] D. P. Jaisi and M. Elimelech, "Single-walled carbon nanotubes exhibit limited transport in soil columns," *Environmental Science & Technology*, vol. 43, no. 24, pp. 9161–9166, 2009.
- [129] D. Kasel, S. A. Bradford, J. Simunek, T. Putz, H. Vereecken, and E. Klumpp, "Limited transport of functionalized multi-walled carbon nanotubes in two natural soils," *Environmental Pollution*, vol. 180, pp. 152–158, 2013.
- [130] Y. Wang, Y. Li, H. Kim, S. L. Walker, L. M. Abriola, and K. D. Pennell, "Transport and retention of fullerene nanoparticles in natural soils," *Journal of Environmental Quality*, vol. 39, no. 6, pp. 1925–1933, 2010.
- [131] D. A. Navarro, R. S. Kookana, J. K. Kirby et al., "Behaviour of fullerenes (C₆₀) in the terrestrial environment: potential release from biosolids-amended soils," *Journal of Hazardous Materials*, vol. 262, pp. 496–503, 2013.

- [132] S. P. Mukherjee, A. R. Gliga, B. Lazzaretto et al., "Graphene oxide is degraded by neutrophils and the degradation products are non-genotoxic," *Nanoscale*, vol. 10, no. 3, pp. 1180–1188, 2018.
- [133] C. Zhang, W. Chen, and P. J. J. Alvarez, "Manganese peroxidase degrades pristine but not surface-oxidized (carboxylated) single-walled carbon nanotubes," *Environmental Science & Technology*, vol. 48, no. 14, pp. 7918–7923, 2014.
- [134] Y. Zhao, B. L. Allen, and A. Star, "Enzymatic degradation of multiwalled carbon nanotubes," *The Journal of Physical Chemistry A*, vol. 115, no. 34, pp. 9536–9544, 2011.
- [135] K. M. Schreiner, T. R. Filley, R. A. Blanchette et al., "White-rot basidiomycete-mediated decomposition of C60 fullerol," *Environmental Science & Technology*, vol. 43, no. 9, pp. 3162–3168, 2009.
- [136] F. Nasser, J. Constantinou, and I. Lynch, "Nanomaterials in the environment acquire an eco-corona impacting their toxicity to daphnia magna—a call for updating toxicity testing policies," *Proteomics*, vol. 20, no. 9, Article ID 1800412, 2020.
- [137] G. Pulido-Reyes, F. Leganes, F. Fernandez-Pinas, and R. Rosal, "Bio-nano interface and environment: a critical review," *Environmental Toxicology and Chemistry*, vol. 36, no. 12, pp. 3181–3193, 2017.
- [138] M. Lundqvist, J. Stigler, T. Cedervall et al., "The evolution of the protein corona around nanoparticles: a test study," *ACS Nano*, vol. 5, no. 9, pp. 7503–7509, 2011.
- [139] V. A. Senapati, K. Kansara, R. Shanker, A. Dhawan, and A. Kumar, "Monitoring characteristics and genotoxic effects of engineered nanoparticle-protein corona," *Mutagenesis*, vol. 32, no. 5, pp. 479–490, 2017.
- [140] M. T. Ekvall, J. Hedberg, I. Odnevall Wallinder, A. Malmendal, L. A. Hansson, and T. Cedervall, "Adsorption of bio-organic eco-corona molecules reduces the toxic response to metallic nanoparticles in *Daphnia magna*," *Scientific Reports*, vol. 11, no. 1, p. 10784, 2021.
- [141] J. Malejko, N. Szymanska, A. Bajguz, and B. Godlewska-Zylkiewicz, "Studies on the uptake and transformation of gold (III) and gold nanoparticles in a water-green algae environment using mass spectrometry techniques," *Journal of Analytical Atomic Spectrometry*, vol. 34, no. 7, pp. 1485–1496, 2019.
- [142] S. M. Briffa, F. Nasser, E. Valsami-Jones, and I. Lynch, "Uptake and impacts of polyvinylpyrrolidone (PVP) capped metal oxide nanoparticles on *daphnia magna*: role of core composition and acquired corona," *Environmental Sciences: Nano*, vol. 5, no. 7, pp. 1745–1756, 2018.
- [143] K. Zhou, Y. Hu, L. Zhang, K. Yang, and D. Lin, "The role of exopolymeric substances in the bioaccumulation and toxicity of Ag nanoparticles to algae," *Scientific Reports*, vol. 6, no. 1, p. 32998, 2016.
- [144] X. Huang, Y. Li, K. Chen et al., "NOM mitigates the phytotoxicity of AgNPs by regulating rice physiology, root cell wall components and root morphology," *Environmental Pollution*, vol. 260, Article ID 113942, 2020.
- [145] K. Kansara, A. Kumar, and A. S. Karakoti, "Combination of humic acid and clay reduce the ecotoxic effect of TiO₂ NPs: a combined physico-chemical and genetic study using zebrafish embryo," *Science of the Total Environment*, vol. 698, Article ID 134133, 2020.
- [146] A. R. Al-Jubory and R. D. Handy, "Uptake of titanium from TiO₂ nanoparticle exposure in the isolated perfused intestine of rainbow trout: nystatin, vanadate and novel CO₂-sensitive components," *Nanotoxicology*, vol. 7, no. 8, pp. 1282–1301, 2013.
- [147] M. Neagu, Z. Piperigkou, K. Karamanou et al., "Protein bio-corona: critical issue in immune nanotoxicology," *Archives of Toxicology*, vol. 91, no. 3, pp. 1031–1048, 2017.
- [148] S. Keshavan, P. Calligari, L. Stella, L. Fusco, L. G. Delogu, and B. Fadeel, "Nano-bio interactions: a neutrophil-centric view," *Cell Death & Disease*, vol. 10, no. 8, p. 569, 2019.
- [149] A. E. Nel, L. Madler, D. Velegol et al., "Understanding biophysicochemical interactions at the nano-bio interface," *Nature Materials*, vol. 8, no. 7, pp. 543–557, 2009.
- [150] Y. Wang, R. Cai, and C. Chen, "The nano-bio interactions of nanomedicines: understanding the biochemical driving forces and redox reactions," *Accounts of Chemical Research*, vol. 52, no. 6, pp. 1507–1518, 2019b.
- [151] T. L. Kirschling, P. L. Golas, J. M. Unrine et al., "Microbial bioavailability of covalently bound polymer coatings on model engineered nanomaterials," *Environmental Science & Technology*, vol. 45, no. 12, pp. 5253–5259, 2011.
- [152] A. Baun, N. B. Hartmann, K. Grieger, and K. O. Kusk, "Ecotoxicity of engineered nanoparticles to aquatic invertebrates: a brief review and recommendations for future toxicity testing," *Ecotoxicology*, vol. 17, no. 5, pp. 387–395, 2008.
- [153] F. R. Khan, K. B. Paul, A. D. Dybowska et al., "Accumulation dynamics and acute toxicity of silver nanoparticles to *daphnia magna* and *Lumbriculus variegatus*: implications for metal modeling approaches," *Environmental Science & Technology*, vol. 49, no. 7, pp. 4389–4397, 2015.
- [154] M. Cozzari, A. C. Elia, N. Pacini et al., "Bioaccumulation and oxidative stress responses measured in the estuarine ragworm (*Nereis diversicolor*) exposed to dissolved, nano- and bulk-sized silver," *Environmental Pollution*, vol. 198, pp. 32–40, 2015.
- [155] F. R. Khan, S. K. Misra, J. Garcia-Alonso et al., "Bioaccumulation dynamics and modeling in an estuarine invertebrate following aqueous exposure to nanosized and dissolved silver," *Environmental Science & Technology*, vol. 46, no. 14, pp. 7621–7628, 2012.
- [156] M. N. Croteau, S. K. Misra, S. N. Luoma, and E. Valsami-Jones, "Silver bioaccumulation dynamics in a freshwater invertebrate after aqueous and dietary exposures to nanosized and ionic Ag," *Environmental Science & Technology*, vol. 45, no. 15, pp. 6600–6607, 2011.
- [157] T. Stoiber, M. N. Croteau, I. Romer, M. Tejamaya, J. R. Lead, and S. N. Luoma, "Influence of hardness on the bioavailability of silver to a freshwater snail after waterborne exposure to silver nitrate and silver nanoparticles," *Nanotoxicology*, vol. 9, no. 7, pp. 918–927, 2015.
- [158] F. R. Khan, A. Laycock, A. Dybowska et al., "Stable isotope tracer to determine uptake and efflux dynamics of ZnO nano- and bulk particles and dissolved Zn to an estuarine snail," *Environmental Science & Technology*, vol. 47, no. 15, pp. 8532–8539, 2013.
- [159] M. N. Croteau, A. D. Dybowska, S. N. Luoma, and E. Valsami-Jones, "A novel approach reveals that zinc oxide nanoparticles are bioavailable and toxic after dietary exposures," *Nanotoxicology*, vol. 5, no. 1, pp. 79–90, 2011.
- [160] K. Wiench, W. Wohlleben, V. Hisgen et al., "Acute and chronic effects of nano- and non-nano-scale TiO₂ and ZnO particles on mobility and reproduction of the freshwater invertebrate *Daphnia magna*," *Chemosphere*, vol. 76, no. 10, pp. 1356–1365, 2009.
- [161] C. M. Zhao and W. X. Wang, "Comparison of acute and chronic toxicity of silver nanoparticles and silver nitrate to

- Daphnia magna*,” *Environmental Toxicology and Chemistry*, vol. 30, no. 4, pp. 885–892, 2011.
- [162] A. H. Ringwood, N. Levi-Polyachenko, and D. L. Carroll, “Fullerene exposures with oysters: embryonic, adult, and cellular responses,” *Environmental Science & Technology*, vol. 43, no. 18, pp. 7136–7141, 2009.
- [163] K. L. Garner, S. Suh, H. S. Lenihan, and A. A. Keller, “Species sensitivity distributions for engineered nanomaterials,” *Environmental Science & Technology*, vol. 49, no. 9, pp. 5753–5759, 2015.
- [164] E. Kadar, O. Dyson, R. D. Handy, and S. N. Al-Subiai, “Are reproduction impairments of free spawning marine invertebrates exposed to zero-valent nano-iron associated with dissolution of nanoparticles?” *Nanotoxicology*, vol. 7, no. 2, pp. 135–143, 2013.
- [165] K. Hund-Rinke, A. Baun, D. Cupi et al., “Regulatory ecotoxicity testing of nanomaterials—proposed modifications of OECD test guidelines based on laboratory experience with silver and titanium dioxide nanoparticles,” *Nanotoxicology*, vol. 10, no. 10, pp. 1442–1447, 2016.
- [166] C. J. Smith, B. J. Shaw, and R. D. Handy, “Toxicity of single walled carbon nanotubes to rainbow trout (*Oncorhynchus mykiss*): respiratory toxicity, organ pathologies, and other physiological effects,” *Aquatic Toxicology*, vol. 82, no. 2, pp. 94–109, 2007.
- [167] R. D. Handy, T. B. Henry, T. M. Scown, B. D. Johnston, and C. R. Tyler, “Manufactured nanoparticles: their uptake and effects on fish—a mechanistic analysis,” *Ecotoxicology*, vol. 17, no. 5, pp. 396–409, 2008.
- [168] C. L. Schultz, A. Wamuch, O. V. Tsyusko et al., “Multi-generational exposure to silver ions and silver nanoparticles reveals heightened sensitivity and epigenetic memory in *Caenorhabditis elegans*,” *Proceedings of the Royal Society B: Biological Sciences*, vol. 283, no. 1832, Article ID 20152911, 2016.
- [169] E. J. Petersen, Q. Huang, and W. J. Weber, “Ecological uptake and depuration of carbon nanotubes by *Lumbricus variegatus*,” *Environmental Health Perspectives*, vol. 116, no. 4, pp. 496–500, 2008.
- [170] Z. Tong, M. Bischoff, L. Nies, B. Applegate, and R. F. Turco, “Impact of fullerene (C60) on a soil microbial community,” *Environmental Science & Technology*, vol. 41, no. 8, pp. 2985–2991, 2007.
- [171] Y. Ge, J. H. Priester, M. Mortimer et al., “Long-term effects of multiwalled carbon nanotubes and graphene on microbial communities in dry soil,” *Environmental Science & Technology*, vol. 50, no. 7, pp. 3965–3974, 2016.
- [172] F. Mouchet, P. Landois, E. Sarremejean et al., “Characterisation and in vivo ecotoxicity evaluation of double-wall carbon nanotubes in larvae of the amphibian *Xenopus laevis*,” *Aquatic Toxicology*, vol. 87, no. 2, pp. 127–137, 2008.
- [173] S. Nations, M. Wages, J. E. Canas, J. Maul, C. Theodorakis, and G. P. Cobb, “Acute effects of Fe₂O₃, TiO₂, ZnO and CuO nanomaterials on *Xenopus laevis*,” *Chemosphere*, vol. 83, no. 8, pp. 1053–1061, 2011.
- [174] J. M. Unrine, O. V. Tsyusko, S. E. Hunyadi, J. D. Judy, and P. M. Bertsch, “Effects of particle size on chemical speciation and bioavailability of copper to earthworms (*Eisenia fetida*) exposed to copper nanoparticles,” *Journal of Environmental Quality*, vol. 39, no. 6, pp. 1942–1953, 2010.
- [175] W. A. Shoults-Wilson, B. C. Reinsch, O. V. Tsyusko, P. M. Bertsch, G. V. Lowry, and J. M. Unrine, “Role of particle size and soil type in toxicity of silver nanoparticles to earthworms,” *Soil Science Society of America Journal*, vol. 75, no. 2, pp. 365–377, 2011.
- [176] T. Gomes, C. G. Pereira, C. Cardoso, J. P. Pinheiro, I. Cancio, and M. J. Bebianno, “Accumulation and toxicity of copper oxide nanoparticles in the digestive gland of *Mytilus galloprovincialis*,” *Aquatic Toxicology*, vol. 118–119, pp. 72–79, 2012.
- [177] G. A. Al-Bairuty, B. J. Shaw, R. D. Handy, and T. B. Henry, “Histopathological effects of waterborne copper nanoparticles and copper sulphate on the organs of rainbow trout (*Oncorhynchus mykiss*),” *Aquatic Toxicology*, vol. 126, pp. 104–115, 2013.
- [178] C. S. Ramsden, T. J. Smith, B. J. Shaw, and R. D. Handy, “Dietary exposure to titanium dioxide nanoparticles in rainbow trout, (*Oncorhynchus mykiss*): no effect on growth, but subtle biochemical disturbances in the brain,” *Ecotoxicology*, vol. 18, no. 7, pp. 939–951, 2009.
- [179] T. Sovova, D. Boyle, K. A. Sloman, C. Vanegas Pérez, and R. D. Handy, “Impaired behavioural response to alarm substance in rainbow trout exposed to copper nanoparticles,” *Aquatic Toxicology*, vol. 152, pp. 195–204, 2014.
- [180] P. L. McNeil, D. Boyle, T. B. Henry, R. D. Handy, and K. A. Sloman, “Effects of metal nanoparticles on the lateral line system and behaviour in early life stages of zebrafish (*Danio rerio*),” *Aquatic Toxicology*, vol. 152, pp. 318–323, 2014.
- [181] E. Cabisco Catala, J. Tamarit Sumalla, and J. Ros Salvador, “Oxidative stress in bacteria and protein damage by reactive oxygen species,” *International Microbiology*, vol. 3, no. 1, pp. 3–8, 2000.
- [182] H. Jang, L. E. Pell, B. A. Korgel, and D. S. English, “Photoluminescence quenching of silicon nanoparticles in phospholipid vesicle bilayers,” *Journal of Photochemistry and Photobiology A: Chemistry*, vol. 158, no. 2–3, pp. 111–117, 2003.
- [183] M. Minghetti and K. Schirmer, “Effect of media composition on bioavailability and toxicity of silver and silver nanoparticles in fish intestinal cells (RTgutGC),” *Nanotoxicology*, vol. 10, no. 10, pp. 1526–1534, 2016.
- [184] S. T. Stern, P. P. Adisheshaiah, and R. M. Crist, “Autophagy and lysosomal dysfunction as emerging mechanisms of nanomaterial toxicity,” *Particle and Fibre Toxicology*, vol. 9, no. 1, p. 20, 2012.
- [185] Y. Matsumura, K. Yoshikata, S. I. Kunisaki, and T. Tsuchido, “Mode of bactericidal action of silver zeolite and its comparison with that of silver nitrate,” *Applied and Environmental Microbiology*, vol. 69, no. 7, pp. 4278–4281, 2003.
- [186] Y. Kai, Y. Komazawa, A. Miyajima, N. Miyata, and Y. Yamakoshi, “60-Fullerene as a novel photoinduced antibiotic,” *Fullerenes, Nanotubes, and Carbon Nanostructures*, vol. 11, no. 1, pp. 79–87, 2003.
- [187] K. H. Park, M. Chhowalla, Z. Iqbal, and F. Sesti, “Single-walled carbon nanotubes are a new class of ion channel blockers,” *Journal of Biological Chemistry*, vol. 278, no. 50, pp. 50212–50216, 2003.
- [188] G. J. Mahler, M. B. Esch, E. Tako et al., “Oral exposure to polystyrene nanoparticles affects iron absorption,” *Nature Nanotechnology*, vol. 7, no. 4, pp. 264–271, 2012.
- [189] J. A. Imlay, “Pathways of oxidative damage,” *Annual Review of Microbiology*, vol. 57, no. 1, pp. 395–418, 2003.
- [190] M. Green and E. Howman, “Semiconductor quantum dots and free radical induced DNA nicking,” *Chemical Communications*, vol. 1, pp. 121–123, 2005.

- [191] I. Nabiev, S. Mitchell, A. Davies et al., "Nonfunctionalized nanocrystals can exploit a cell's active transport machinery delivering them to specific nuclear and cytoplasmic compartments," *Nano Letters*, vol. 7, no. 11, pp. 3452–3461, 2007.
- [192] B. O. Dabbousi, J. Rodriguez-Viejo, F. V. Mikulec et al., "(CdSe)/ZnS core-shell quantum dots: synthesis and characterization of a size series of highly luminescent nanocrystallites," *The Journal of Physical Chemistry B*, vol. 101, no. 46, pp. 9463–9475, 1997.
- [193] S. Kim, B. Fisher, H. J. Eisler, and M. Bawendi, "Type-II quantum dots: CdTe/CdSe (core/shell) and CdSe/ZnTe (core/shell) heterostructures," *Journal of the American Chemical Society*, vol. 125, no. 38, pp. 11466–11467, 2003.
- [194] S. Takenaka, K. Yamashita, M. Takagi, T. Hatta, A. Tanaka, and O. Tsuge, "Study of the DNA interaction with water-soluble cationic fullerene derivatives," *Chemistry Letters*, vol. 28, no. 4, pp. 319–320, 1999.
- [195] X. Zhao, A. Striolo, and P. T. Cummings, "C60 binds to and deforms nucleotides," *Biophysical Journal*, vol. 89, no. 6, pp. 3856–3862, 2005.
- [196] S. Takenaka, K. Yamashita, M. Takagi, T. Hatta, and O. Tsuge, "Photo-induced DNA cleavage by water-soluble cationic fullerene derivatives," *Chemistry Letters*, vol. 28, no. 4, pp. 321–322, 1999.
- [197] G. Wakefield, M. Green, S. Lipscomb, and B. Flutter, "Modified titania nanomaterials for sunscreen applications-reducing free radical generation and DNA damage," *Materials Science and Technology*, vol. 20, no. 8, pp. 985–988, 2004.
- [198] D. Kuhnel, K. Scheffler, P. Wellner et al., "Comparative evaluation of particle properties, formation of reactive oxygen species and genotoxic potential of tungsten carbide-based nanoparticles *in vitro*," *Journal of Hazardous Materials*, vol. 227–228, pp. 418–426, 2012.
- [199] P. V. AshaRani, G. Low Kah Mun, M. P. Hande, and S. Valiyaveetil, "Cytotoxicity and genotoxicity of silver nanoparticles in human cells," *ACS Nano*, vol. 3, no. 2, pp. 279–290, 2009.
- [200] L. Gonzalez, D. Lison, and M. Kirsch-Volders, "Genotoxicity of engineered nanomaterials: a critical review," *Nanotoxicology*, vol. 2, no. 4, pp. 252–273, 2008.
- [201] S. J. H. Soenen, N. Nuytten, S. F. De Meyer, S. C. De Smedt, and M. De Cuyper, "High intracellular iron oxide nanoparticle concentrations affect cellular cytoskeleton and focal adhesion kinase-mediated signaling," *Small*, vol. 6, no. 7, pp. 832–842, 2010.
- [202] A. Millaku, D. Drobne, M. Torkar, S. Novak, M. Remskar, and Z. Pipan-Tkalec, "Use of scanning electron microscopy to monitor nanofibre/cell interaction in digestive epithelial cells," *Journal of Hazardous Materials*, vol. 260, pp. 47–52, 2013.
- [203] R. Bacchetta, E. Moschini, N. Santo et al., "Evidence and uptake routes for Zinc oxide nanoparticles through the gastrointestinal barrier in *Xenopus laevis*," *Nanotoxicology*, vol. 8, no. 7, pp. 728–744, 2014.
- [204] E. Frohlich, "Cellular targets and mechanisms in the cytotoxic action of non-biodegradable engineered nanoparticles," *Current Drug Metabolism*, vol. 14, no. 9, pp. 976–988, 2013.
- [205] T. Mashino, K. Okuda, T. Hirota, M. Hirobe, T. Nagano, and M. Mochizuki, "Inhibition of *E. coli* growth by fullerene derivatives and inhibition mechanism," *Bioorganic & Medicinal Chemistry Letters*, vol. 9, no. 20, pp. 2959–2962, 1999.
- [206] T. Mashino, D. Nishikawa, K. Takahashi et al., "Antibacterial and antiproliferative activity of cationic fullerene derivatives," *Bioorganic & Medicinal Chemistry Letters*, vol. 13, no. 24, pp. 4395–4397, 2003.
- [207] A. Thill, O. Zeyons, O. Spalla et al., "Cytotoxicity of CeO₂ nanoparticles for *Escherichia coli*. Physico-chemical insight of the cytotoxicity mechanism," *Environmental Science & Technology*, vol. 40, no. 19, pp. 6151–6156, 2006.
- [208] T. C. King-Heiden, P. N. Wicinski, A. N. Mangham et al., "Quantum dot nanotoxicity assessment using the zebrafish embryo," *Environmental Science & Technology*, vol. 43, no. 5, pp. 1605–1611, 2009.
- [209] J. Duan, Y. Yu, Y. Li, Y. Yu, and Z. Sun, "Cardiovascular toxicity evaluation of silica nanoparticles in endothelial cells and zebrafish model," *Biomaterials*, vol. 34, no. 23, pp. 5853–5862, 2013.
- [210] K. T. Kim, T. Zaikova, J. E. Hutchison, and R. L. Tanguay, "Gold nanoparticles disrupt zebrafish eye development and pigmentation," *Toxicological Sciences*, vol. 133, no. 2, pp. 275–288, 2013.
- [211] A. P. Francis, T. Devasena, S. Ganapathy, V. R. Palla, P. B. Murthy, and S. Ramaprabhu, "Multi-walled carbon nanotube-induced inhalation toxicity: recognizing nano bisdemethoxy curcumin analog as an ameliorating candidate," *Nanomedicine: Nanotechnology, Biology and Medicine*, vol. 14, no. 6, pp. 1809–1822, 2018.
- [212] M. Durga, S. Nathiya, and T. Devasena, "Protective role of fenugreek leaf extract and Quercetin against petrol exhaust nanoparticle induced lipid peroxidation and oxidative stress in rat erythrocytes *in vitro*," *Asian Journal of Pharmaceutical Sciences and Clinical Research*, vol. 8, no. 1, pp. 237–241, 2015.
- [213] M. Durga and T. Devasena, "Cardioprotective potential of quercetin against diesel or petrol exhaust nanoparticle induced toxicity: a prospective *in vitro* pharmacological study in H9C2 cells," *Asian Journal of Pharmaceutical and Clinical Research*, vol. 14, no. 14, pp. 139–144, 2021.
- [214] Q. Wang, S. D. Ebbs, Y. Chen, and X. Ma, "Trans-generational impact of cerium oxide nanoparticles on tomato plants," *Metallomics*, vol. 5, no. 6, p. 753, 2013.
- [215] W. Du, J. L. Gardea-Torresdey, R. Ji et al., "Physiological and biochemical changes imposed by CeO₂ nanoparticles on wheat: a life cycle field study," *Environmental Science & Technology*, vol. 49, no. 19, pp. 11884–11893, 2015.
- [216] J. H. Priester, Y. Ge, R. E. Mielke et al., "Soybean susceptibility to manufactured nanomaterials with evidence for food quality and soil fertility interruption," *Proceedings of the National Academy of Sciences of the United States of America*, vol. 109, no. 37, pp. E2451–E2456, 2012.
- [217] L. Zhao, Y. Sun, J. A. Hernandez-Viezcas et al., "Monitoring the environmental effects of CeO₂ and ZnO nanoparticles through the life cycle of corn (*Zea mays*) plants and *in situ* μ -XRF mapping of nutrients in kernels," *Environmental Science & Technology*, vol. 49, no. 5, pp. 2921–2928, 2015.
- [218] J. D. Judy, D. H. McNear, C. Chen et al., "Nanomaterials in biosolids inhibit nodulation, shift microbial community composition, and result in increased metal uptake relative to bulk/dissolved metals," *Environmental Science & Technology*, vol. 49, no. 14, pp. 8751–8758, 2015.
- [219] W. Du, Y. Sun, R. Ji, J. Zhu, J. Wu, and H. Guo, "TiO₂ and ZnO nanoparticles negatively affect wheat growth and soil enzyme activities in agricultural soil," *Journal of Environmental Monitoring*, vol. 13, no. 4, p. 822, 2011.

- [220] H. Zhu, J. Han, J. Q. Xiao, and Y. Jin, "Uptake, translocation, and accumulation of manufactured iron oxide nanoparticles by pumpkin plants," *Journal of Environmental Monitoring*, vol. 10, no. 6, p. 713, 2008.
- [221] Y. Feng, X. Cui, S. He et al., "The role of metal nanoparticles in influencing arbuscular mycorrhizal fungi effects on plant growth," *Environmental Science & Technology*, vol. 47, no. 16, pp. 9496–9504, 2013.
- [222] L. Yang and D. J. Watts, "Particle surface characteristics may play an important role in phytotoxicity of alumina nanoparticles," *Toxicology Letters*, vol. 158, no. 2, pp. 122–132, 2005.
- [223] S. Zhang, R. Deng, D. Lin, and F. Wu, "Distinct toxic interactions of TiO₂ nanoparticles with four coexisting organochlorine contaminants on algae," *Nanotoxicology*, vol. 11, no. 9–10, pp. 1115–1126, 2017.
- [224] L. Li, M. Sillanpaa, and E. Schultz, "Influence of titanium dioxide nanoparticles on cadmium and lead bioaccumulations and toxicities to *Daphnia magna*," *Journal of Nanoparticle Research*, vol. 19, no. 6, p. 223, 2017.
- [225] N. K. Geitner, W. Zhao, F. Ding, W. Chen, and M. R. Wiesner, "Mechanistic insights from discrete molecular dynamics simulations of pesticide-nanoparticle interactions," *Environmental Science & Technology*, vol. 51, no. 15, pp. 8396–8404, 2017.
- [226] D. Lin, X. Tian, F. Wu, and B. Xing, "Fate and transport of engineered nanomaterials in the environment," *Journal of Environmental Quality*, vol. 39, no. 6, pp. 1896–1908, 2010.
- [227] R. Khan, M. A. Inam, S. Khan, D. R. Park, and I. T. Yeom, "Interaction between persistent organic pollutants and ZnO NPs in synthetic and natural waters," *Nanomaterials*, vol. 9, no. 3, p. 472, 2019.
- [228] M. Wang, B. Gao, D. Tang, H. Sun, X. Yin, and C. Yu, "Effects of temperature on aggregation kinetics of graphene oxide in aqueous solutions," *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, vol. 538, pp. 63–72, 2018.
- [229] K. Sun, S. Dong, Y. Sun et al., "Graphene oxide-facilitated transport of levofloxacin and ciprofloxacin in saturated and unsaturated porous media," *Journal of Hazardous Materials*, vol. 348, pp. 92–99, 2018.
- [230] A. N. Afrooz, D. Das, C. J. Murphy, P. Vikesland, and N. B. Saleh, "Co-transport of gold nanospheres with single-walled carbon nanotubes in saturated porous media," *Water Research*, vol. 99, pp. 7–15, 2016.
- [231] E. Sahle-Demessie, C. Han, A. Zhao, B. Hahn, and H. Grecsek, "Interaction of engineered nanomaterials with hydrophobic organic pollutants," *Nanotechnology*, vol. 27, no. 28, Article ID 284003, 2016.
- [232] B. Bhaduri, T. Polubesova, and B. Chefetz, "Interactions of organic dye with Ag- and Ce-nano-assemblies: influence of dissolved organic matter," *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, vol. 577, pp. 683–694, 2019.
- [233] X. Wu, W. Wang, and L. Zhu, "Enhanced organic contaminants accumulation in crops: mechanisms, interactions with engineered nanomaterials in soil," *Environmental Pollution*, vol. 240, pp. 51–59, 2018.
- [234] Q. Zhao, Y. Li, X. Chai et al., "Interaction of nano carbon particles and anthracene with pulmonary surfactant: the potential hazards of inhaled nanoparticles," *Chemosphere*, vol. 215, pp. 746–752, 2019.
- [235] J. Jia, F. Li, S. Zhai et al., "Susceptibility of over-weight mice to liver injury as a result of the ZnO nanoparticle-enhanced liver deposition of Pb²⁺," *Environmental Science & Technology*, vol. 51, no. 3, pp. 1775–1784, 2017.
- [236] A. T. Besha, Y. Liu, C. Fang, D. N. Bekele, and R. Naidu, "Assessing the interactions between micropollutants and nanoparticles in engineered and natural aquatic environments," *Critical Reviews in Environmental Science and Technology*, vol. 50, no. 2, pp. 135–215, 2019.
- [237] P. Li, X. Zou, X. Wang et al., "A preliminary study of the interactions between microplastics and citrate-coated silver nanoparticles in aquatic environments," *Journal of Hazardous Materials*, vol. 385, Article ID 121601, 2020.
- [238] L. Cai, L. He, S. Peng, M. Li, and M. Tong, "Influence of titanium dioxide nanoparticles on the transport and deposition of microplastics in quartz sand," *Environmental Pollution*, vol. 253, pp. 351–357, 2019.
- [239] B. Huang, Z. B. Wei, L. Y. Yang, K. Pan, and A. J. Miao, "Combined toxicity of silver nanoparticles with hematite or plastic nanoparticles toward two freshwater algae," *Environmental Science & Technology*, vol. 53, no. 7, pp. 3871–3879, 2019.
- [240] J. D. Judy, J. M. Unrine, and P. M. Bertsch, "Evidence for biomagnification of gold nanoparticles within a terrestrial food chain," *Environmental Science & Technology*, vol. 45, no. 2, pp. 776–781, 2011.
- [241] J. M. Unrine, W. A. Shoultz-Wilson, O. Zhurbich, P. M. Bertsch, and O. V. Tsyusko, "Trophic transfer of Au nanoparticles from soil along a simulated terrestrial food chain," *Environmental Science & Technology*, vol. 46, no. 17, pp. 9753–9760, 2012.
- [242] J. Hawthorne, R. De la Torre Roche, B. Xing et al., "Particle-size dependent accumulation and trophic transfer of cerium oxide through a terrestrial food chain," *Environmental Science & Technology*, vol. 48, no. 22, pp. 13102–13109, 2014.
- [243] R. De la Torre-Roche, A. Servin, J. Hawthorne et al., "Terrestrial trophic transfer of bulk and nanoparticle La₂O₃ does not depend on particle size," *Environmental Science & Technology*, vol. 49, no. 19, pp. 11866–11874, 2015.
- [244] S. Majumdar, J. Trujillo-Reyes, J. A. Hernandez-Viezcas, J. C. White, J. R. Peralta-Videa, and J. L. Gardea-Torresdey, "Cerium biomagnification in a terrestrial food chain: influence of particle size and growth stage," *Environmental Science & Technology*, vol. 50, no. 13, pp. 6782–6792, 2016.