Transformation Driven Assessment of Environmental Behaviour and Ecotoxicity of Engineered Nanoparticles

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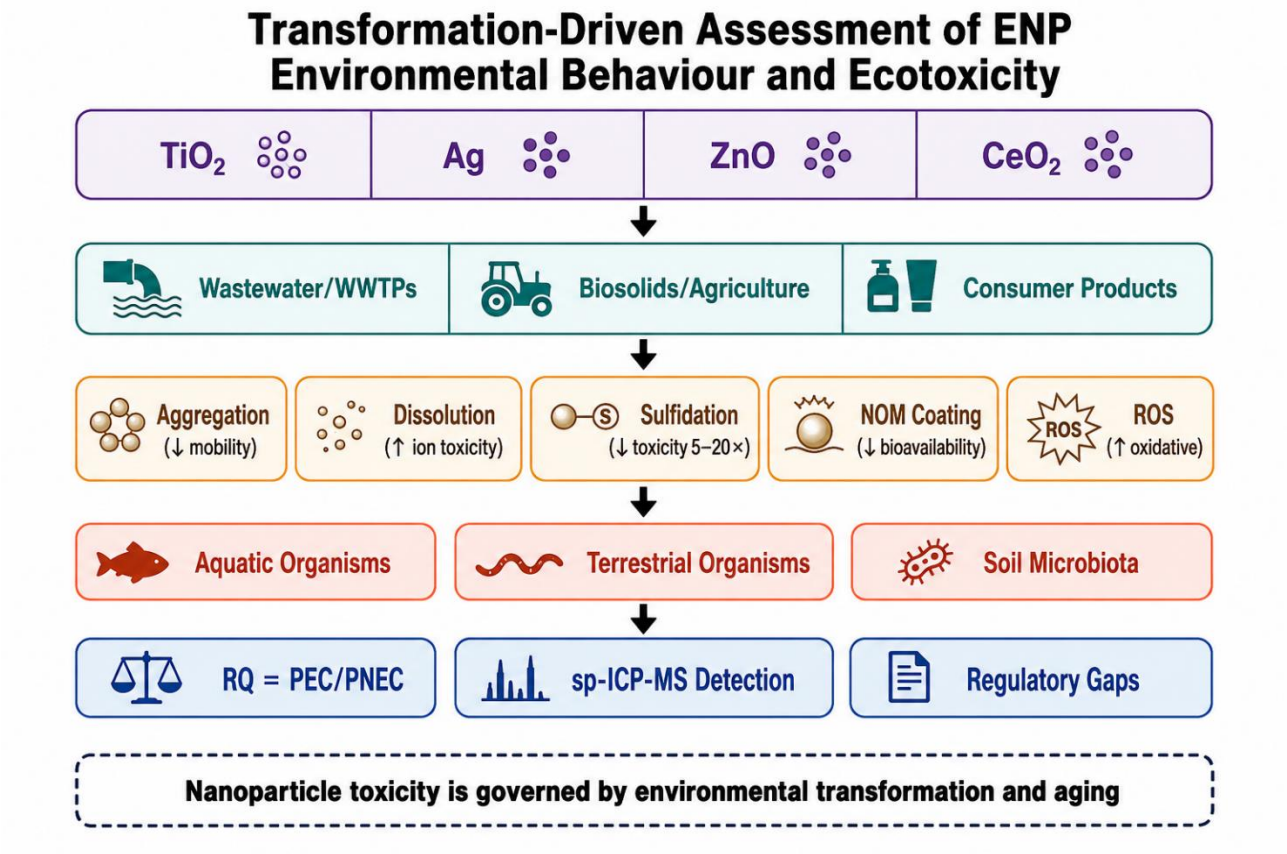
ABSTRACT

Engineered nanoparticles (ENPs) are widely used in industry, agriculture and consumer products. Their residues are gradually released into the environment via wastewater treatment plants and biosolid application. This review critically analyses the environmental fate, transformation processes, ecotoxicological effects, analytical detection methods and risk assessment frameworks of the major ENP types: TiO_2, Ag, ZnO and CeO_2. ENPs are subject to physicochemical transformations (aggregation, dissolution, sulfidation and coating with natural organic matter (NOM)) after environmental release that can significantly affect their mobility, bioavailability and toxicity. Predicted environmental concentrations are 0.003–16 µg/L in surface water for Ag and TiO_2

respectively and 0.01–10 mg/kg in soil from biosolid applications. Ecotoxicological evidence indicates high sensitivity among aquatic organisms. 48-hour LC50 values of 2-40 µg/L have been reported for Daphnia magna and EC50 values of 0.05-5 mg/L have been reported for freshwater algae. Sulfidation reduces the toxicity of silver nanoparticles by 5–20 times, showing that environmental transformation can significantly reduce risks. Single-particle ICP-MS has improved environmental detection, but differentiating engineered nanoparticles from natural nanoparticles remains a key analytical challenge. Existing risk assessment schemes do not take bioavailability changes related to transformations into account and environmental quality standards specific to nanomaterials are lacking. The review emphasises the importance of transformation-aware risk models, standardised testing procedures and safe-by-design approaches for environmentally sustainable nanotechnology development.

KEYWORDS: *Engineered nanoparticles, Environmental transformation, Ecotoxicity, Single-particle ICP-MS, Risk assessment*

Graphical Abstract:



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1. INTRODUCTION

The rapid growth of nanotechnology over the past two decades has led to the large-scale production and use of engineered nanoparticles (ENPs) in diverse sectors such as medicine, agriculture, energy, electronics and consumer goods (Vance *et al.*, 2015). The worldwide production of nanomaterial is still growing significantly, e.g. the production of titanium dioxide (TiO_2) is more than 10,000 tonnes per year and the production of silver nanoparticles (AgNPs) is 500 tonnes per year (Piccinno *et al.*, 2012). These particles (1–100 nm) have unique properties of physicochemical nature, such as high surface area-to-volume ratio, quantum effects and increased reactivity, which make them different from their bulk counterparts (Joudeh and Linke, 2022; Nel *et al.*, 2006). Commercial applications include drug delivery and diagnostics for pharmaceuticals (Mitchell *et al.*, 2021), nano-fertilizers and pesticides (Kah *et al.*, 2018), solar cells and batteries (Li *et al.*, 2018), antibacterial clothes and cosmetics (Venkatesan *et al.*, 2017), and environmental remediation catalysts (Gawande *et al.*, 2016). The most commonly produced ENP categories include metal oxides (TiO_2, ZnO, CeO_2), metal-based nanoparticles (Ag, Au, Cu), and carbon nanomaterial (CNTs, graphene) (Bundschuh *et al.*, 2018).

However, the widespread utility of ENPs has raised environmental concerns for unintentional release, persistence and ecological impacts as integration into commercial products increases (Gottschalk *et al.*, 2009). ENPs are released to the environment during manufacturing emissions, wastewater discharge, product wear and washing and landfill disposal (Keller *et al.*, 2013). Wastewater treatment plants (WWTPs) are important point sources that transfer 70–99% of influent nanoparticles into sewage sludge and discharge the remaining 1–30% into receiving waters (Westerhoff *et al.*, 2011). The direct terrestrial ENP exposure pathway is via application of biosolids to agricultural land (Kiser *et al.*, 2009). Once released, ENPs undergo several physicochemical transformations such as aggregation, dissolution, surface coating by natural organic matter (NOM) and chemical speciation changes, which strongly influence their mobility, bioavailability and toxicity (Levard *et al.*, 2013; Lowry *et al.*, 2012). Due to their persistence in sediments and soils, they are considered emerging contaminants (Nowack *et al.*, 2012). According to models, the concentration of TiO_2 in surface water

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is predicted to be 0.7-16 µg/L and for Ag it is 0.003-0.5 µg/L, while soils may have concentrations up to 0.1-100 mg/kg from biosolids application (Gottschalk *et al.*, 2013; Sun *et al.*, 2016).

There are still considerable knowledge gaps in understanding how ENPs behave and impact ecosystems under realistic exposure conditions (Nel *et al.*, 2013). Laboratory studies typically use nanoparticles at high concentrations in simplified media, with poor representation of complex environmental matrices that undergo continuous transformation (Atofarati *et al.*, 2026; Bondarenko *et al.*, 2013). Long-term field monitoring is scarce and analytical detection at environmentally relevant concentrations (ng/L to µg/L) while differentiating engineered from natural nanoparticles is technically challenging (von der Kammer *et al.*, 2012; Laborda *et al.*, 2013; Pérez-López *et al.*, 2026). The presence of multiple mechanisms, such as particle-specific effects and toxicity of dissolved ions, also complicates the assessment of toxicity, and is not adequately captured by conventional risk assessment frameworks (Hou et al., 2013). Inconsistent reporting of characterisation and the lack of standardised testing protocols continue to hinder the comparability between studies and reproducibility of data (Petersen et al., 2015).

1.1. Methodology of Review

In this review, a structured literature search was performed in three major scientific databases: Scopus, Web of Science and Google Scholar. The following keyword combinations were used for searches performed in January 2026: *"engineered nanoparticles AND environmental fate", "nanoparticle transformation AND ecotoxicity", "TiO_2 OR AgNPs OR ZnO OR CeO_2 AND risk assessment", "nanoparticle AND aquatic toxicity OR terrestrial toxicity", and "single-particle ICP-MS AND environmental detection"*. The search was limited to peer-reviewed journal articles published between 2005 and 2025.

Inclusion criteria were: (a) peer-reviewed studies on environmental behaviour, transformation, toxicity or risk assessment of ENPs; (b) studies that reported quantitative data on concentrations, toxicity endpoints or transformation rates; and (c) studies that were conducted in environmental or environmentally-relevant systems. Exclusion criteria were: (a) research that was purely biomedical

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or clinical, (b) non-environmental systems that were not ecologically relevant, and (c) research that lacked quantitative data or adequate characterisation.

Initial database searches identified approximately 200 studies after removal of duplicates. Following abstract screening for inclusion/exclusion criteria, 100 studies were assessed for eligibility at full text level. After full text screening, 70 studies were included for detailed synthesis. This review has been structured as a critical narrative review rather than a systematic review. No formal study-quality scoring system was applied but the strength of evidence is discussed qualitatively throughout. Data extracted from each study included: *type of ENP, environmental compartment, concentration range, transformation mechanism, toxicity endpoint, species tested, and analytical method*.

1.2 Integrated Conceptual Framework of ENP Behaviour and Ecotoxicity

The environmental behaviour of ENPs is the product of interrelated processes that connect release pathways, physicochemical transformations, bioavailability and biological responses across trophic levels. This review adopts an integrated conceptual framework that systematically links these processes to provide an integrated interpretation of ENP fate and ecotoxicological effects in environmental systems.

ENPs are mainly released into air, water, and soil through industrial emissions, wastewater discharges, and biosolid application. They subsequently undergo a series of physicochemical transformations including aggregation, dissolution, sulfidation and eco-corona formation by NOM interactions that alter particle size, surface chemistry and chemical speciation and control environmental transport and persistence. Significantly, these transformations determine the bioavailability of nanoparticles, which is the main factor leading to ecotoxicological effects at different trophic levels. Aggregation and NOM coating usually decrease bioavailability and toxicity, whereas dissolution can release of bioavailable metal ions that increase toxicity especially for Ag and ZnO. This framework highlights that ecotoxicity is not an intrinsic property of the material but an emergent feature of environmentally aged forms under specific exposure conditions, a perspective

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that exposes the limitations of standard static risk assessment approaches. The interconnections between these processes are shown in Figure 1.

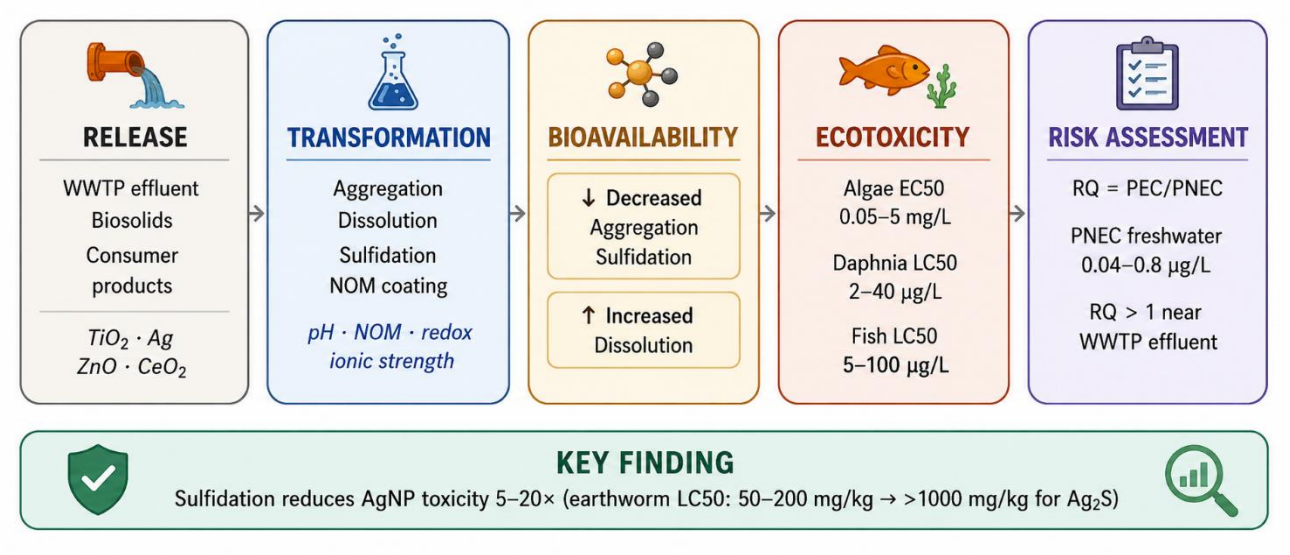


Figure 1. Integrated conceptual framework linking ENP release, transformation, bioavailability, and ecotoxicity

1.3 Scope and Objectives:

Against this background, the present review critically synthesizes current knowledge on the environmental fate, transformation mechanisms, ecotoxicological effects, analytical detection methods, and risk assessment frameworks for the four most widely studied ENP types: TiO_2, Ag, ZnO, and CeO_2. Specifically, the review addresses the following questions: (i) Which transformation pathways dominate under different environmental conditions? (ii) When does transformation reduce ecological risk and when does it increase risk? (iii) Which risk assessment models are limited by their treatment of ENPs as static materials? and (iv) Where is the evidence strong, moderate, or contradictory? By focusing on environmentally aged rather than pristine nanoparticles, and by integrating findings from recent field studies alongside laboratory data, this review aims to provide a transformation-driven assessment that advances beyond descriptive synthesis toward a critically evaluated, evidence-weighted framework for ENP risk assessment.

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2. ENVIRONMENTAL FATE AND TRANSFORMATION

2.1 Release Pathways and Environmental Distribution

Engineered nanoparticles are released into the environment through multiple pathways during their life cycle (Keller *et al.*, 2013). The primary pathway for atmospheric release during synthesis and handling is via airborne emissions from manufacturing facilities, although levels of exposure at regulated sites have been mitigated via improved occupational controls (Froggett *et al.*, 2014; Lee *et al.*, 2026). Industrial process wastewater contributes to aquatic releases (Oladipo, 2026; Westerhoff *et al.*, 2011). Consumer products are significant diffuse sources, sunscreens contribute 1–4 μg/L TiO_2 and ZnO to recreational waters (Labille *et al.*, 2010), antimicrobial textiles release 0.1–20% of their silver content in washing (Benn and Westerhoff, 2008), and exterior paints release photocatalytic TiO_2 slowly through weathering at about 10–100 ng/cm^2/year (Al-Kattan *et al.*, 2013). ENPs are directly applied to soil and crops through the agricultural use of nano-pesticides and nano-fertilizers (Islam, 2025; Kah *et al.*, 2018). Veterinary applications add to loads in manure (Dimkpa and Bindraban, 2018).

Wastewater treatment plants (WWTPs) are important control points, where nanoparticles are concentrated by aggregation-enhanced sedimentation and biosorption. Typical activated sludge processes remove 70–99% of influent nanoparticles into sewage sludge, at concentrations 100–1000 times higher than in the effluent (Cavazzoli *et al.*, 2023). Biosolids contain silver in the range of 1-100 mg/kg dry weight and TiO_2 in the range of 1-6 mg/kg. The remaining 1–30% of nanoparticles are released in effluent at concentrations of 0.01–10 μg/L for silver and 5–50 μg/L for TiO_2 (Keller *et al.*, 2013; Westerhoff *et al.*, 2011). The primary pathway for terrestrial ENP exposure is through land application of biosolids. Approximately 50-60% of sewage sludge in the EU and USA is used as a soil amendment. Mass balance modelling predicts that repeated applications of biosolids over 10–100 years would result in accumulation of 0.1–5 mg/kg silver in agricultural topsoil (Sun *et al.*, 2016). Table 1 presents predicted and measured nanoparticle concentrations across key environmental compartments.

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Table 1. Predicted and Measured Nanoparticle Concentrations in Environmental Compartments

ENP Type	Surface Water PEC (µg/L)	Surface Water MEC (µg/L)	Sediment (mg/kg)	Soil PEC (mg/kg)	Biosolids (mg/kg)	Reference
TiO_2	0.7-16	0.7-5.4	10-500	0.5–5	1000-6000	(Gondikas *et al.*, 2014; Gottschalk *et al.*, 2013; Sun *et al.*, 2016)
Ag	0.003-0.5	0.003-0.5	0.05-5	0.01-2	1-100	(Azimzada *et al.*, 2017; Gottschalk *et al.*, 2013; Sun *et al.*, 2016)
ZnO	0.1-3	0.1-3	5-50	0.1-5	100-1000	(Gottschalk *et al.*, 2013; Sun *et al.*, 2016)
CeO_2	0.01-1	0.01-1	0.5-10	0.01-1	10-200	(Gottschalk *et al.*, 2013; Sun *et al.*, 2016)

PEC = predicted environmental concentration (model-derived); MEC = measured environmental concentration (field-measured). Surface water MEC values are from field studies; sediment and soil PEC values are primarily model-derived from (Gottschalk *et al.*, 2013; Sun *et al.*, 2016). *TiO_2 surface water MEC based on seasonal monitoring at Old Danube recreational lake* (Gondikas *et al.*, 2014). *TiO_2 soil PEC reflects 10-year cumulative biosolids application scenario* (Sun *et al.*, 2016)

2.2 Transformation Processes

ENPs undergo physical, chemical, and biological transformations after release into the environment that significantly alter their properties (Feculak *et al.*, 2025; Padmanabhan *et al.*, 2026). Aggregation is through electrostatic and van der Waals forces, resulting in the formation of larger particle clusters, the rate of which is dependent on ionic strength, pH and natural organic matter (NOM) concentration (Azimzada *et al.*, 2017; Hotze *et al.*, 2010; Lowry *et al.*, 2012). AgNPs and TiO_2 nanoparticles are relatively stable in freshwater at low ionic strength (1 mM) and rapidly aggregate at seawater salinity (500–700 mM) (Keller *et al.*, 2010). The resulting aggregates display reduced mobility and increased sedimentation rates, leading to a transfer of nanoparticles from the water column to sediments (Quik *et al.*, 2012). Chemical transformations highly influence properties and toxicity of ENPs. Dissolution refers to the release of metal ions from the particle surfaces. Silver undergoes oxidative dissolution

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188 ($4Ag^0 + O_2 + 4H^+ \rightarrow 4Ag^+ + 2H_2O$) with rates increasing with smaller particle sizes, higher
189 temperatures and lower pH. Studies have shown that 10 nm AgNPs dissolved 2-3 times faster than
190 50 nm particles (Hosny *et al.*, 2026; Liu and Hurt, 2010) and that surface coatings of AgNPs were
191 important in the rate of dissolution, with citrate-coated AgNPs dissolving faster than PVP-coated
192 particles. ZnO dissolves rapidly, particularly in acidic conditions (pH 4-6) releasing Zn^{2+} ions, which
193 are the main driving force for its toxicity whereas TiO_2 and CeO_2 show limited dissolution in
194 environmental pH ranges (David et al., 2012). For silver, transformation by sulfidation is of particular
195 importance in anaerobic environments such as WWTPs and anoxic sediments. The process occurs in
196 two steps. First, Ag^0 is oxidised to Ag^+; then it precipitates as silver sulphide (Ag_2S) when reacting
197 with dissolved sulphide (HS^-/H_2S) (Levard et al., 2013). The overall transformation is represented as:
198 $Ag^0 \rightarrow Ag^+ \rightarrow Ag_2S$. The formed Ag_2S is poorly soluble and much less toxic than AgNPs or dissolved
199 Ag^+. Field studies show that 50–90% of silver in sewage sludge is present as Ag_2S (Kiser et al., 2009;
200 Levard et al., 2013). Table 2 summarises the key transformation processes and their effects on ENP
201 mobility and toxicity.

202 **Table 2. Transformation processes and their effects on nanoparticle mobility and toxicity**

Process	Mechanism	Key Factors	Effect on Mobility	Effect on Toxicity	Reference
Aggregation	Electrostatic, van der Waals forces	Ionic strength (+), pH, NOM (±)	Decreased (sedimentation)	Decreased (reduced bioavailability)	(Keller *et al.*, 2010; Hotze *et al.*, 2010)
Dissolution	Oxidative/protonation release of ions	Particle size (−), pH (−), O_2 (+), coating type	Increased (ionic species)	Increased Ag^+, Zn^{2+} release (Ag, ZnO)	(David *et al.*, 2012; Liu and Hurt, 2010)
Sulfidation	$Ag^0 \rightarrow Ag^+ \rightarrow Ag_2S$ (two-step)	Redox potential, sulfide availability	Decreased (low solubility)	Greatly decreased (5–20×)	(Levard *et al.*, 2013)
NOM coating	Surface adsorption, eco-corona formation	NOM concentration, pH, ENP surface charge	Variable	Decreased	(Stankus *et al.*, 2011) Grillo *et al.*, 2015)
Photocatalysis	UV-induced ROS generation	UV light intensity, O_2, water depth	Variable	Increased (TiO_2, ZnO)	(Miller *et al.*, 2010)

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NOM = natural organic matter; (+) positive correlation; (−) negative correlation; (±) variable effect.

**Photocatalysis applies only in sunlit surface waters; this process is negligible in sediments, soils, and subsurface environments where UV penetration is absent.*

NOM modification of the surface leads to the formation of an eco-corona which alters colloidal stability and biological interactions (Chakraborty and Lynch, 2025; Stankus *et al.*, 2011). The NOM coating generally increases the particle stability and could decrease the bioavailability by covering the reactive surface sites (Grillo et al., 2015). In surface waters, TiO_2 and ZnO can photocatalytically react under UV irradiation, forming ROS, which can enhance toxicity to aquatic organisms (Miller et al., 2010). This photocatalytic effect is most relevant in sunlit surface waters and does not apply to sediments, soils, or subsurface environments where UV penetration is negligible. The simultaneous occurrence of multiple transformation processes results in time-dependent changes in ENP properties, generally resulting in environmentally aged particles that are larger, surface-coated and display significantly different toxicity profiles compared to pristine laboratory materials (Afrooz *et al.*, 2013; Grillo *et al.*, 2015; Martínez *et al.*, 2020). Figure 2 illustrates the distribution of ENPs in the environment and main transformation pathways of ENPs in aquatic and terrestrial compartments.

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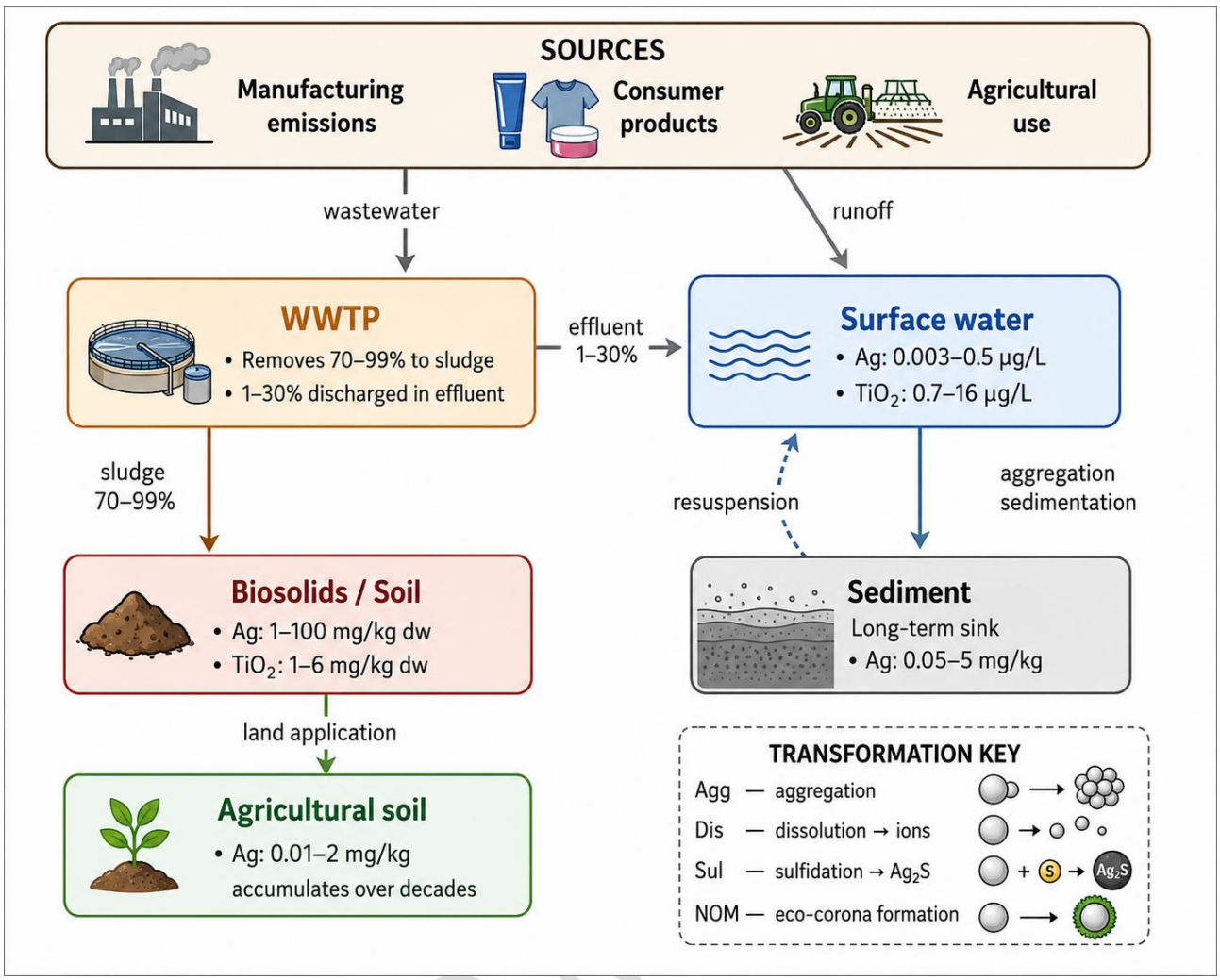


Figure 2. Environmental distribution and transformation pathways of engineered nanoparticles across compartments.

Figure 2 illustrates release pathways from WWTPs and consumer products, partitioning among water, sediment and soil compartments, and key physicochemical transformation processes (Ridoy and Supto, 2025) (aggregation, dissolution, sulfidation, NOM coating and photocatalysis). Compartment boundaries are conceptual transitions; concentration values are indicative and are based on modelled and measured data in Table. 1.

The contrasting effects of different transformation processes (aggregation reducing mobility, dissolution increasing ionic toxicity) highlight the inherent complexity of ENP environmental behaviour and the limitations of generalisation of risk from single-process studies. Most of the experimental studies focus on single transformation pathways in isolation, resulting in simple

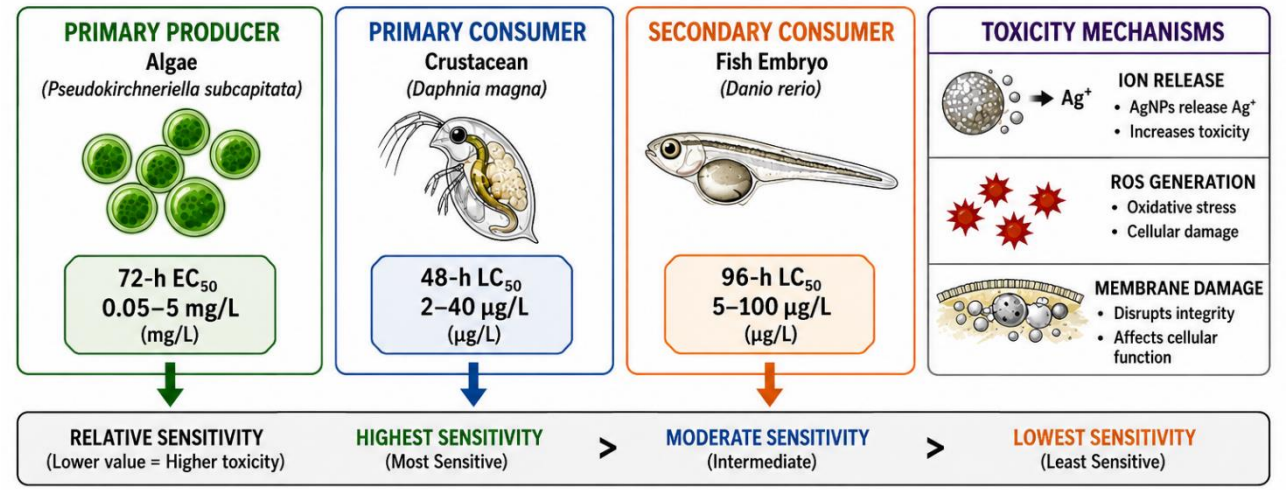
interpretations. Further studies should incorporate multiple concurrent pathways of transformation and directly relate physicochemical changes to biological responses to enhance the accuracy of environmental risk predictions.

3. ECOTOXICOLOGICAL IMPACTS

3.1 Aquatic Toxicity

Water column dispersion and sediment accumulation expose aquatic organisms to ENPs (Bondarenko et al., 2013). Algae are the principal producers and the basic entrance point to aquatic food webs. Green algae (Pseudokirchneriella subcapitata) exposed to AgNPs have 72-hour EC50 values of 0.05–5 mg/L, several orders of magnitude lower than bulk silver, which demonstrates the great bioavailability of nano-silver and dissolved Ag^+ ions (Levard et al., 2013). Mechanisms of toxicity include oxidative breakdown and release of Ag^+ ions and direct contact of particles with membranes leading to disruption of nutrient absorption. TiO_2 has a very low intrinsic toxicity (EC50 ranges 1-50 mg/L) but its photocatalytic ROS production under UV irradiation significantly increases its effects in surface waters (Miller et al., 2010). High algal toxicity of ZnO (EC50 = 0.02–2 mg/L) is mostly due to fast Zn^{2+} dissolution (David et al., 2012). Species sensitivity varies; marine diatoms are often more tolerant than freshwater green algae. Chronic exposure to low levels (≤10 μg/L AgNPs) can negatively affect photosynthesis and lead to reproductive impacts over several generations (Bondarenko et al., 2013). Figure 3 presents AgNP toxicity ranges across aquatic trophic levels based on data from Table 3.

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LINK TO TABLE 3 – TOXICITY VALUES

Organism	Algae (*P. subcapitata*)	Crustacean (*D. magna*)	Fish Embryo (*D. rerio*)	Endpoint & Exposure Duration
AgNPs	0.05–5 mg/L	2–40 µg/L	5–100 µg/L	72-h EC_{50} / 48-h LC_{50} / 96-h LC_{50}
TiO_2 NPs	1–50 mg/L	1–20 mg/L	10–200 mg/L	72-h EC_{50} / 48-h LC_{50} / 96-h LC_{50}
ZnO NPs	0.02–2 mg/L	0.5–10 mg/L	2–20 mg/L	72-h EC_{50} / 48-h LC_{50} / 96-h LC_{50}

Figure 3. AgNP aquatic toxicity ranges across trophic levels.

Aquatic invertebrates are critical in food web dynamics and nutrient cycling. Daphnia magna has 48-h LC50 values of 2–40 µg/L for AgNPs with toxicity mechanisms including gut obstruction by ingestion of particles, Ag^+ uptake through the carapace and oxidative stress (Bondarenko et al., 2013; Levard et al., 2013). Reproduction was adversely affected after chronic exposure at 1 µg/L in 21-day experiments. Bivalves such as mussels and oysters are able to bioaccumulate nanoparticles with bioconcentration factors (BCFs) ranging from 100 to 10,000 depending on particle type and speciation (Judy *et al.*, 2011; Le *et al.*, 2025). Sediment-dwelling invertebrates are relatively less sensitive, with LC50s generally 10-100 times greater than similar water-column exposures, because of reduced bioavailability in sediment matrices (Quik et al., 2012). Figure 3 shows that the sensitivity of algae and crustaceans is higher than that of fish at the same exposure concentrations. Values include EC50/LC50 ranges for Pseudokirchneriella subcapitata (algae, 72-h EC50: 0.05–5 mg/L), Daphnia magna (crustacean, 48-h LC50: 2–40 µg/L), and Danio rerio embryo (fish, 96-h LC50: 5–100 µg/L) from laboratory investigations with pure AgNPs. Table 3 presents ecotoxicological data for key aquatic organisms across trophic levels.

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Table 3. Aquatic organism toxicity endpoints for common engineered nanoparticles by trophic level

Organism	Species	ENP	Duration	Endpoint	Test Endpoint Detail	Concentration	Mechanism	Reference
Algae	*P. subcapitata*	Ag	72 h	EC50	Growth inhibition	0.05–5 mg/L	Ag^+, ROS, membrane disruption	Levard *et al.*, 2013
Algae	*P. subcapitata*	TiO_2	96 h	EC50	Growth inhibition	1–50 mg/L	ROS (light-dependent)	Miller *et al.*, 2010
Algae	*P. subcapitata*	ZnO	72 h	EC50	Growth inhibition	0.02–2 mg/L	Zn^{2+} dissolution	David *et al.*, 2012
Crustacean	*D. magna*	Ag	48 h	LC50	Mortality	2–40 µg/L	Gut blockage, Ag^+, oxidative stress	Bondarenko *et al.*, 2013; Levard *et al.*, 2013
Crustacean	*D. magna*	Ag	21 d	NOEC	Reproduction inhibition	0.5–5 µg/L	Chronic ion toxicity	Bondarenko *et al.*, 2013
Fish	*D. rerio* (embryo)	Ag	96 h	LC50	Mortality/hatching	5–100 µg/L	Ion toxicity, developmental delay	Bondarenko *et al.*, 2013
Fish	*D. rerio* (adult)	Ag	14–28 d	NOEC	Gill damage, behaviour	1–10 µg/L	Oxidative stress, neurotoxicity	Bondarenko *et al.*, 2013

EC50 = median effective concentration; LC50 = median lethal concentration; NOEC = no observed effect concentration. All values represent pristine nanoparticle exposures in laboratory conditions; environmentally aged or sulfidized forms exhibit substantially reduced toxicity. AgNP EC50 for algae based on Levard et al. (2013); values reflect dissolved Ag^+ contribution.

Fish are vertebrate models for ENP toxicity evaluation. AgNPs have LC50 values of 5–100 µg/L in zebrafish (Danio rerio) embryo-larval experiments, which also reveal developmental consequences such as hatching delay and spinal abnormalities (Bondarenko et al., 2013). Adult fish subjected to 1–10 µg/L AgNPs showed gill damage, oxidative stress indicators (increased catalase and superoxide dismutase activity) and neurotoxicity as shown by changes in swimming behaviour. Bioaccumulation occurs mainly in gills, liver and intestinal tissue with distribution patterns depending on size and surface qualities of the particles (Kumar *et al.*, 2026; Oros, 2025). Trophic transfer studies have shown that most ENPs undergo trophic dilution, rather than biomagnification in aquatic food chains, **unlike the biomagnification commonly associated with persistent organic pollutants**(Judy et al., 2011).

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However, this pattern is material and food-chain specific, with different forms of ENP and exposure circumstances showing differing transfer patterns. This topic requires further exploration before broad generalisations can be drawn.

3.2 Terrestrial Toxicity

ENPs are mostly introduced to terrestrial ecosystems by biosolid land application, atmospheric deposition and pesticide use. Soil microbial populations are particularly sensitive indicators of the influence of ENPs due to their essential role in decomposing organic matter and performing nutrient cycles. AgNPs (10–100 mg/kg) diminish bacterial diversity, inhibit important soil enzyme activities such as dehydrogenase (20–50%), urease (15–40%), and phosphatase (10–30%), and depress nitrification (30–60%) (Ivask et al., 2014). The results from Ivask et al. (2014) are from E. coli liquid culture studies and impacts of the soil matrix including lower bioavailability due to sorption and sulfidation would indicate that toxicity thresholds in real soils are likely to be higher than would be inferred from liquid-culture values. Longer term investigations (several months duration) often report partial functional recovery especially in organic-rich soils where sulfidation progressively transforms AgNPs to Ag_2S (Levard et al., 2013; Whitley et al., 2013). Effects on mycorrhizal fungal communities can be detected at 50–100 mg/kg (Ivask et al., 2014). Aged biosolid-derived Ag_2S is shown to have far less microbial toxicity than pure AgNPs, but total abrogation of effects has not been established at concentrations above 100 mg/kg (Whitley et al., 2013).

Table 4 represents terrestrial organism toxicity and ecosystem effects.

Table 4. Terrestrial Organism Toxicity and Ecosystem Effects

Target	System	ENP	Concentration	Duration	Effect	Magnitude	Reference
Bacteria†	Agricultural soil	Ag	10-100 mg/kg	28 d	Dehydrogenase	20-50% decrease	(Ivask *et al.*, 2014)
Bacteria†	Agricultural soil	Ag	1-10 mg/kg	28 d	Nitrification	30-60% decrease	(Ivask *et al.*, 2014)
Bacteria	Biosolids soil	Ag_2S	10-100 mg/kg	180 d	Microbial biomass	Significantly reduced effect	(Levard *et al.*, 2013;

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Target	System	ENP	Concentration	Duration	Effect	Magnitude	Reference
							Whitley *et al.*, 2013)
Plant	*T. aestivum*	Ag	50-500 mg/kg	14 d	Germination	10-40% reduction	(Cornelis *et al.*, 2014)
Plant	*L. sativa*	ZnO	100-1000 mg/kg	7 d	Root elongation	20-70% reduction	(Cornelis *et al.*, 2014; David *et al.*, 2012)
Earthworm	*E. fetida*	Ag	50-200 mg/kg	28 d	LC50	50-200 mg/kg	(Whitley *et al.*, 2013)
Earthworm	*E. fetida*	Ag_2S	100-1000 mg/kg	28 d	LC50	>1000 mg/kg	(Whitley *et al.*, 2013)
Collembola	*F. candida*	Ag	10-100 mg/kg	28 d	Reproduction	40-80% reduction	(Cornelis *et al.*, 2014)

LC50 = median lethal concentration. †Bacteria toxicity values from Ivask et al. (2014) were derived from E. coli liquid culture experiments; actual soil toxicity thresholds are expected to be higher due to reduced bioavailability in soil matrices. Ag_2S biosolids data from Whitley et al. (2013): significantly reduced but not fully eliminated toxicity at ≥100 mg/kg.

Soil fauna are important organisms for decomposition and maintaining soil structure. Eisenia fetida (earthworm) had 28-day LC50 values of 50–200 mg/kg for pristine AgNPs but above 1000 mg/kg for Ag_2S, indicating a distinct transformation-mediated reduction in toxicity of 5–20-fold (Whitley et al., 2013). Folsomia candida (Collembola/springtail) reproduction is reduced by 40-80% at 10-100 mg/kg AgNPs suggesting sublethal reproductive effects occur at lethal concentrations (Cornelis et al., 2014). Field studies of biosolid amended agricultural soils generally show little effect on crop yield and soil function for realistic ENP concentrations below 10 mg/kg but are highly dependent on soil type, organic matter content, extent of sulfidation, duration of exposure and endpoint sensitivity (Alsayaydeh *et al.*, 2024; Levard *et al.*, 2013; Whitley *et al.*, 2013). Taken together, these results reveal that the toxicity of nanoparticles is not an intrinsic attribute of the material but one that is dynamically regulated by environmental transformation processes and exposure conditions, a finding with direct consequences for the design and application of risk assessment frameworks.

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4. DETECTION AND RISK ASSESSMENT

4.1 Analytical Detection Methods

The detection and characterisation of ENPs in complex environmental matrices remains a significant analytical challenge, because target concentrations range from ng/L to µg/L in water, and from µg/kg to mg/kg in solid matrices, together with the background interference from natural colloids and organic matter. Direct visualisation of particles with sub-nanometer resolution is possible with transmission electron microscopy (TEM) to determine particle size, shape and crystallinity whereas scanning electron microscopy coupled with energy-dispersive X-ray spectroscopy (SEM-EDS) allows surface imaging and elemental mapping (Ellis (Bradford) *et al.*, 2026; von der Kammer *et al.*, 2012; Laborda *et al.*, 2013). Critical disadvantages of both procedures include labour-intensive sample preparation, possibility of preparation artefacts, small sample volumes, and, significantly, the inability to distinguish engineered nanoparticles from morphologically similar natural colloids.

Single-particle inductively coupled plasma mass spectrometry (sp-ICP-MS) represents a major step forward in the environmental analysis of nanoparticles, enabling the simultaneous quantification of particle number concentration, size distribution and dissolved ion concentration from the same sample (Laborda *et al.*, 2013; Meng *et al.*, 2023; Villarruel *et al.*, 2026). In sp-ICP-MS analysis, a diluted suspension is injected into the ICP-MS where individual particles produce discrete signal pulses above the dissolved ion background. Signal strength is translated to particle mass and equivalent spherical dimension. The method has been used for AgNPs, TiO_2, CeO_2 and Au in wastewater effluent, surface water and soil extracts (Azimzada *et al.*, 2017; Montaño *et al.*, 2016). Size detection limits are element dependent, ranging from about 15–20 nm for Ag up to 50–100 nm for TiO_2, and particle number concentration detection limits are around 10^5–10^6 particles/mL (equivalent to 10^{11}–10^{12} particles/L) for 60-nm AgNPs under typical instrument conditions (Laborda et al., 2013; Montaño et al., 2016). Instruments with higher sensitivity, such as ICP-TOF-MS, can broaden the size detection range to smaller particles. Limitations are matrix effects which need to be calibrated

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carefully, density assumptions which lead to inaccuracy in size predictions, and the inability to differentiate between designed and naturally occurring nanoparticles of the same elemental makeup. Emerging methods like multi-element fingerprinting by ICP-MS/MS and machine learning classification of sp-ICP-MS signal signatures have the potential to improve engineered/natural distinction (von der Kammer et al., 2012). Table 5 presents a comparative review of analytical approaches for environmental nanoparticle identification.

Table 5. Analytical Techniques for Environmental Nanoparticle Detection

Technique	Information	Size Range	Detection Limit	Advantages	Limitations	Reference
TEM	Morphology, size, crystallinity	1-1000 nm	Variable (sample-dependent)	Sub-nanometer resolution, crystal structure	Labor-intensive, artifacts, low throughput, no ENP/natural ID	(Laborda *et al.*, 2013)
SEM-EDS	Morphology, elemental composition	5-10,000 nm	Variable	Surface imaging, elemental mapping	Lower resolution than TEM, sample prep required	(Laborda *et al.*, 2013)
DLS	Hydrodynamic diameter, PDI	1-6000 nm	0.1–1 mg/L (minimum for reliable sizing)†	Fast, simple, non-destructive	Unsuitable for polydisperse samples, no elemental ID, intensity-biased	(Hotze *et al.*, 2010)
sp-ICP-MS	Particle size, number concentration, dissolved ions	15–1000 nm‡	10^5–10^6 particles/mL	Quantitative, element-specific, simultaneous size and concentration	Size LOD element-dependent, no ENP/natural ID, density assumptions	(Laborda *et al.*, 2013; Montaño *et al.*, 2016)
AF4-ICP-MS	Size-resolved elemental composition	1-1000 nm	µg/L range	Size fractionation, broad size range	Time-consuming, sample dilution required	(von der Kammer *et al.*, 2012)

PDI = polydispersity index; LOD = limit of detection; DLS = dynamic light scattering; AF4 = asymmetric flow field-flow fractionation; ENP = engineered nanoparticle. †DLS measures intensity fluctuations to derive hydrodynamic diameter; it does not quantify particle number concentration. The value given represents the minimum sample concentration for reliable size measurement. ‡sp-ICP-MS size detection limits are element-dependent: approximately 15–20 nm for Ag and 50–100 nm for TiO_2 under standard instrument conditions; higher-sensitivity instruments (e.g., ICP-TOF-MS) can extend detection to smaller sizes.

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Asymmetric flow field-flow fractionation coupled with ICP-MS (AF4-ICP-MS) allows size-resolved elemental analysis over a wide size range and is particularly useful for polydisperse environmental samples when DLS performance is insufficient (von der Kammer et al., 2012). A common problem for all approaches is to distinguish designed from naturally occurring nanoparticles, which may be indistinguishable in elemental composition and overlapping size ranges. The lack of recognised reference materials for environmental matrices further limits the validation and standardisation of analytical methods amongst laboratories. The development of approved standards, upgrade instruments and expand data processing methods are essential to overcome these analytical limits and provide credible environmental monitoring data.

4.2 Risk Assessment and Regulation

The ecological risk evaluation of ENPs is based on a standard risk quotient (RQ) approach, where the predicted environmental concentration (PEC), is compared with the predicted no-effect concentration (PNEC). RQ = PEC/PNEC and RQ > 1 shows the possible ecological danger (Gottschalk et al., 2013). However, application of this approach to nanomaterials is difficult because of their particular physicochemical features, dynamic transformation behaviour and insufficient toxicity data (Hou et al., 2013; Nel et al., 2013). The exposure estimate is based on material flow models which consider production volumes, product consumption, release rates and environmental partitioning (Gottschalk et al., 2013). Probabilistic models incorporating uncertainty distributions show reasonable agreement (within one order of magnitude) with measured concentrations for TiO_2 and Ag in the well-studied locations (Azimzada et al., 2017; Gondikas et al., 2014). A major source of uncertainty is bioavailability. Aggregated and environmentally aged nanoparticles are less bioavailable than pristine particles (Ellis (Bradford) *et al.*, 2026). Transformation to less toxic forms especially Ag_2S via sulfidation greatly reduces bioavailability, although this is only partially incorporated into existing models (Grillo et al., 2015; Levard et al., 2013). The hazard evaluation is based on multi-species toxicity data to develop species sensitivity distributions (SSDs) and to create PNEC values that are protective for 95% of the species (Petersen et al., 2015). SSDs require large quantities of high quality

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data and the majority of research available are based on pristine nanoparticles at concentrations higher than environmental values, causing systematic bias. For AgNPs, the most robust body of evidence allows derivation of PNEC values of 0.04−0.8 µg/L for freshwater utilising SSD approach (Bondarenko *et al.*, 2013; Chen *et al.*, 2024). These values are compared to AgNP surface water PEC values of 0.003–0.5 µg/L, suggesting that risk quotients might approach or surpass 1 in WWTP-receiving waters, warranting closer regulatory attention (Gottschalk et al., 2013; Sun et al., 2016). Figure 4 presents the transformation-informed risk assessment framework showing how key transformation processes modify PEC, bioavailability, and RQ relative to static assessments.

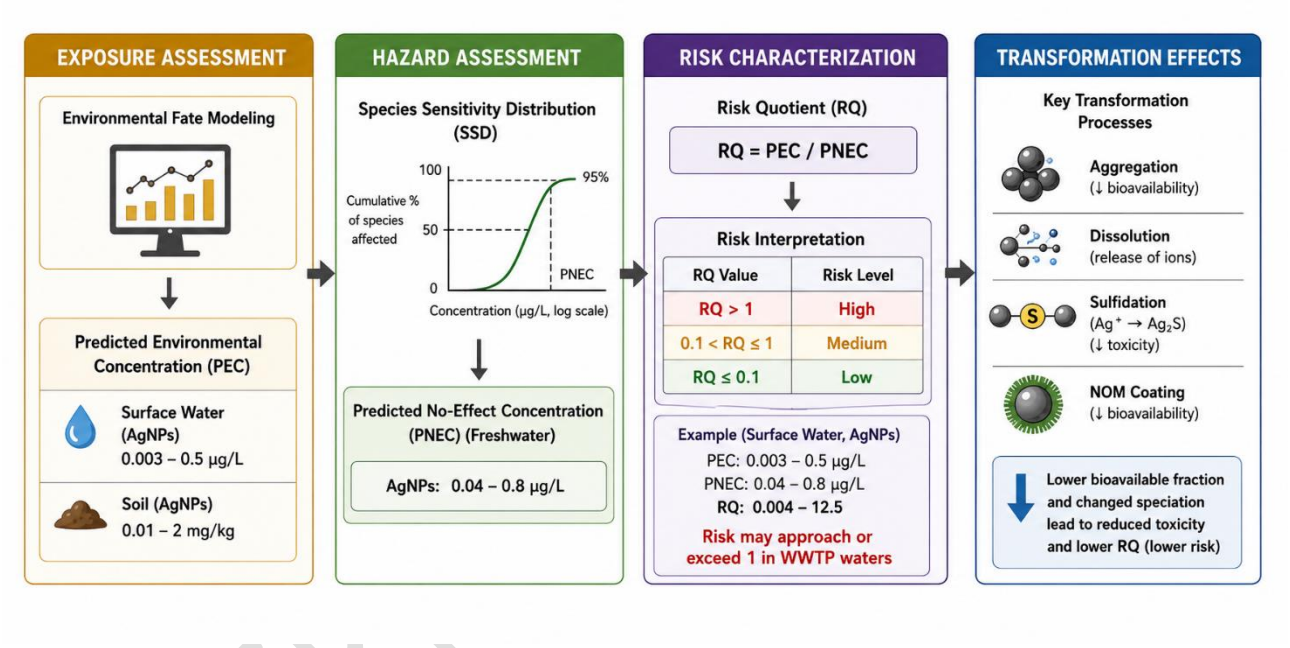


Figure 4: Analytical Detection and Risk Assessment Framework

Figure 4 shows a transformation-based risk assessment paradigm for engineered nanoparticles (ENPs). The framework illustrates how important transformation processes, aggregation, dissolution, sulfidation and NOM coating, change effective PEC (by changing bioavailable fraction), shift PNEC (by changing speciation and toxicity) and thus alter risk quotient (RQ = PEC/PNEC) relative to assessments based on pristine nanoparticle properties. Values are indicative and derived from the data in Tables 1, 3 and 6. Table 6 presents indicative risk quotients for each ENP type across freshwater and soil compartments using PEC and PNEC ranges from Tables 1 and 3.

Table 6. Indicative Risk Quotients (RQ) for ENPs in Environmental Compartments:

ENP	Compartment	PEC Range	PNEC	RQ Range	Risk Indication	Reference
Ag	Freshwater	0.003–0.5 µg/L	0.04–0.8 µg/L	0.004–12.5	Low to high — RQ > 1 near WWTP effluents	Bondarenko *et al.*, 2013; Gottschalk *et al.*, 2013
TiO_2	Freshwater	0.7–16 µg/L	1–10 µg/L	0.07–16	Low to high — RQ > 1 in impacted systems	Gottschalk *et al.*, 2013; Petersen *et al.*, 2015
ZnO	Freshwater	0.1–3 µg/L	0.1–1 µg/L	0.1–30	Low to high — dissolution increases risk	Gottschalk *et al.*, 2013; Bondarenko *et al.*, 2013
CeO_2	Freshwater	0.01–1 µg/L	1–10 µg/L	0.001–1	Generally low	Gottschalk *et al.*, 2013
Ag	Agricultural soil	0.01–2 mg/kg	0.1 mg/kg†	0.1–20	Moderate to high after biosolid accumulation	Sun *et al.*, 2016; Cornelis *et al.*, 2014
ZnO	Agricultural soil	0.1–5 mg/kg	1 mg/kg†	0.1–5	Low to moderate	Sun *et al.*, 2016; Cornelis *et al.*, 2014

RQ = PEC/PNEC; RQ > 1 indicates potential ecological risk. All values are indicative and based on PEC ranges from Table 1 and PNEC estimates from species sensitivity distributions. †Soil PNEC values are provisional estimates based on available ecotoxicological data; nano-specific soil quality standards have not been formally established. RQ ranges reflect the full PEC and PNEC uncertainty ranges; central estimates are lower. Transformation processes (sulfidation, aggregation) would reduce effective PEC and thus RQ in aged systems.

Globally, there are many regulatory systems for nanomaterials. REACH (the European Union's law) requires detailed physicochemical characterisation of compounds generated in quantities greater than 1 tonne per year and established a separate nano-specific registration obligation in 2018. In the United States, nanoparticles are regulated by the Toxic Substances Control Act (TSCA) via case-by-case Significant New Use Rules (Hunt *et al.*, 2025). OECD has revised 15 test criteria to make them applicable to nanomaterials. However, there remains a critical gap: no jurisdiction has yet established nano-specific environmental quality standards (EQS) that distinguish between nanoparticulate, dissolved, and bulk forms that are substantially different in bioavailability and toxicity (Chaudhari *et*

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al., 2025; Petersen *et al.*, 2015). There is limited postmarket environmental monitoring, leaving actual environmental ENP concentrations largely uncharacterised.

The safe-by-design solutions are a proactive way to incorporate environmental issues at the development stage of nanomaterials (Chaudhari *et al.*, 2025; Grillo *et al.*, 2015; Maynard, 2015). Critical strategies include (a) replacing highly toxic elements with less-harmful elements (e.g., iron-based nanoparticles versus silver for antimicrobial applications) (b) controlling particle size and surface chemistry to reduce environmental and biological persistence (c) applying degradable or reactive-site-blocking coatings to reduce bioavailability and (d) designing ENPs to rapidly transform to environmentally benign end-products (e.g., designing ZnO to dissolve to bioavailable Zn^{2+} , an essential plant nutrient, rather than persisting as toxic nanoparticles). Green synthesis approaches with plant extracts and benign solvents give lower-impact alternatives to standard chemical synthesis methods. Life-cycle assessment (LCA) frameworks offer a means for a full assessment of environmental consequences from raw material extraction through end-of-life, facilitating holistic sustainability assessment of nanomaterial products (Babu *et al.*, 2025; Hicks, 2025; Ramadan *et al.*, 2024).

5. Future Perspectives in ENP Environmental Research

Although substantial progress has been made in understanding the environmental fate and ecotoxicological effects of ENPs, numerous key knowledge gaps persist that hinder robust risk assessment and regulatory decisions. This section outlines priority research directions required to shift the field from a descriptive, laboratory-based assessment, to a dynamic, transformation-aware and field-validated knowledge of ENP environmental risk.

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Priority 1: Develop certified environmentally aged ENP reference materials.

The most common constraint across all of the domains evaluated is the use of pure, as-synthesized nanoparticles in laboratory toxicity and fate studies. In real-world environmental systems, ENPs undergo constant transformation including aggregation, dissolution, sulfidation and NOM coating which profoundly changes their properties hours to days after release. Future research should use pre-aged or transformation-simulated ENP suspensions that represent field-relevant speciation. The development of certified reference materials for important ENPs (Ag, TiO_2, ZnO, CeO_2) in their environmentally transformed forms (e.g. Ag_2S, NOM-coated TiO_2) within significant environmental matrices will enable cross laboratory comparability and standardised risk testing.

The scope of this review paperis restricted to metal and metal oxide ENPs (TiO_2, Ag, ZnO, CeO_2) for which sufficient environmental fate and ecotoxicological data exist to support a transformation-driven synthesis. Carbon-based nanomaterials (CNTs, graphene, fullerenes) are outside the scope of this review.

Priority 2: Implement long-term field monitoring.

Most ecotoxicological data available are from short-term (days to weeks) laboratory exposures at concentrations orders of magnitude exceeding ambient values. Chronic toxicity data at environmentally relevant doses (ng/L to µg/L in water; µg/kg to low mg/kg in soil) at across multiple trophic levels are still sparse for all four evaluated ENP classes. Long-term field monitoring studies in WWTP receiving waters and biosolid amended agricultural soils are particularly needed to confirm model predictions and demonstrate real-world ecosystem-level effects.

Priority 3: Integrate transformation kinetics into risk assessment frameworks.

Existing risk assessment methods consider ENPs as static materials with fixed properties and fixed PEC and PNEC values that do not account for **time-dependent changes in speciation.** The transformation-aware risk assessment approach would include: (a) transformation kinetics in environmental compartments (WWTP, freshwater, sediment, soil); (b) compartment-specific

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bioavailability fractions for pristine vs. aged forms of ENP; and (c) dynamic RQ calculation, which varies with progressing transformation. Table 7 provides a transformation-by-compartment evidence matrix for the four ENPs assessed, scoring the strength of available evidence (strong, moderate, or weak) for each transformation pathway in each compartment.

Priority 4: Develop harmonised analytical methodologies and certified reference materials.

Harmonised analytical techniques across laboratories are needed for reliable environmental monitoring. No standardised calibration strategies exist for sp-ICP-MS systems at present. Element-specific size detection limits (Ag: 15-20 nm; TiO_2: 50-100 nm) are higher than the size range of most environmentally significant ENP fractions. Key steps towards the generation of reproducible and comparable monitoring datasets include the development of certified reference materials for environmental matrices, the harmonisation of international sp-ICP-MS calibration protocols, and the advancement of emerging techniques including multi-element ICP-MS/MS fingerprinting and machine learning classification of particle signal signatures.

Priority 5: Develop nano-specific environmental quality standards (EQS).

No jurisdiction has yet defined formal EQS that discriminate between the nanoparticulate, dissolved and bulk forms of metals in environmental compartments. For freshwater AgNPs, the available PNEC values obtained from SSDs (0.04–0.8 µg/L) and PEC estimates (0.003–0.5 µg/L) show that RQ values may approach or surpass 1 in the vicinity of WWTP discharges, but there is no regulatory trigger to require action. Science-based regulatory thresholds for nano-specific EQS adjusted for bioavailability data from transformation would promote the use of safe-by-design production techniques.

Priority 6: Develop a systematic transformation-by-ENP evidence synthesis.

One major limitation noted in this research is the absence of a structured cross-comparison of transformation behaviour and corresponding risk alterations across the four ENP types and relevant environmental compartments. Such synthesis would enable evidence-based prioritisation of research investment and regulatory attention. Table 7 summarises the proposed evidence-confidence matrix of

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existing knowledge strength for each transformation-compartment combination across TiO_2, Ag, ZnO and CeO_2. The matrix illustrates that sulfidation of Ag in WWTP and sludge systems and dissolution of Ag and ZnO in freshwater are the best characterised transformation-risk linkages presently available, while transformation behaviour of CeO_2 across all compartments and effects of photocatalysis in non-surface-water environments remain poorly evidenced and warrant prioritised investigation.

Table 7. Transformation-by-ENP-Compartment Evidence Confidence Matrix:

Transformation	Compartment	Ag	TiO_2	ZnO	CeO_2
Sulfidation	WWTP/Sludge	Strong ✓	Not applicable	Moderate	Weak
Sulfidation	Freshwater sediment	Moderate	Not applicable	Weak	Weak
Dissolution	Freshwater	Strong ✓	Weak	Strong ✓	Weak
Aggregation	Freshwater	Strong ✓	Strong ✓	Moderate	Moderate
NOM coating	Freshwater	Moderate	Moderate	Moderate	Weak
Photocatalysis	Surface water	Weak	Strong ✓	Moderate	Weak
Transformation → risk reduction	Soil/Biosolids	Strong ✓	Weak	Moderate	Weak

Strong: multiple independent studies with consistent findings; Moderate: limited studies or conflicting results; Weak: single study or largely uncharacterised. ✓ indicates transformation reduces ecological risk; entries without ✓ indicate transformation increases or has variable effect on risk. "Not applicable" indicates the transformation pathway does not apply to this ENP type.

Addressing these six priorities through coordinated international research programmes, standardised testing frameworks and transformation-aware regulatory approaches will help to facilitate a transition from current static, material-centric risk assessment to a dynamic, system-level understanding of ENP environmental behaviour and hence responsible and sustainable governance of nanotechnology.

6. CONCLUSIONS

WWTP effluents and biosolid land application are the main routes via which engineered nanoparticles are introduced into aquatic and terrestrial systems, with surface water concentrations in the range of

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0.003–16 μg/L and soil concentrations in the range of 0.01–10 mg/kg, depending on ENP type. However, concentration statistics alone are not adequate to describe environmental danger. Following release into the environment, ENP mobility, bioavailability and toxicity are continuously modified by transformation processes such as aggregation, dissolution, sulfidation and NOM coating. The key conclusion from this review is that ENP toxicity is not an inherent material characteristic but an emergent property of environmentally aged forms under certain exposure conditions. Sulfidation reduces silver nanoparticle toxicity by 5–20-fold, demonstrated by earthworm LC50 shifting from 50–200 mg/kg for pristine AgNPs to >1000 mg/kg for Ag_2S, while dissolution of Ag and ZnO releases bioavailable metal ions that substantially increase toxicity relative to aggregated or sulfidized forms.

Aquatic organisms are generally more vulnerable to ENP exposure than terrestrial organisms, with deleterious effects found at concentrations of μg/L. AgNP LC50 values of 2-40 μg/L for Daphnia magna and risk quotients at or above 1 for WWTP discharges indicate ecologically significant exposure in impacted systems. Terrestrial effects are highly dependent on soil parameters, depth of transformation and duration of exposure; field investigations in biosolid-amended soils suggest generally modest effects below 10 mg/kg but results vary widely with soil type and organic matter concentration. Trophic transfer studies suggest dilution rather than biomagnification for most examined ENP types and food chains, although this trend needs to be verified in a material-specific and chain-specific way before broad generalisation.

Advances in analytical techniques especially sp-ICP-MS have greatly enhanced the detection of ENPs in complex environmental matrices. However, critical unresolved limitations include size detection limits of 15–20 nm for Ag and 50–100 nm for TiO_2, and the inability to differentiate engineered and natural nanoparticles. There are no jurisdictional nano-specific environmental quality standards distinguishing nanoparticulate, dissolved and bulk forms, a regulatory gap that erodes evidence-based risk management. Closing these gaps needs transformation-integrated risk models with dynamic RQ calculations, long-term field monitoring programmes, certified environmentally aged reference materials and internationally harmonised analytical and regulatory standards. Responsible and

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sustainable governance of nanotechnology will be further supported by implementing safe-by-design methodologies and life-cycle assessment frameworks, minimising ecological risks.

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REFERENCES

Afrooz, A.R.M.N., Sivalapalan, S.T., Murphy, C.J., Hussain, S.M., Schlager, J.J. and Saleh, N.B. (2013), "Spheres vs. rods: The shape of gold nanoparticles influences aggregation and deposition behavior", *Chemosphere*, Vol. 91 No. 1, pp. 93–98, doi: https://doi.org/10.1016/j.chemosphere.2012.11.031.

Al-Kattan, A., Wichser, A., Vonbank, R., Brunner, S., Ulrich, A., Zuin, S. and Nowack, B. (2013), "Release of TiO2 from paints containing pigment-TiO2 or nano-TiO2 by weathering", *Environmental Science: Processes & Impacts*, The Royal Society of Chemistry, Vol. 15 No. 12, pp. 2186–2193, doi: 10.1039/C3EM00331K.

Alsayaydeh, J., Yusof, M.F., Magenthiran, M., Hamzah, R., Mustaffa, I. and Herawan, S. (2024), "Empowering crop cultivation: harnessing internet of things for smart agriculture monitoring", *International Journal of Electrical and Computer Engineering (IJECE)*, Vol. 14, p. 6023, doi: 10.11591/ijece.v14i5.pp6023-6035.

Atofarati, E.O., Ajuka, L.O. and Enweremadu, C.C. (2026), "Life cycle assessment of nanomaterials and laboratory consumables: toward safer and sustainable nanotechnology", *Journal of Nanoparticle Research*, Vol. 28 No. 4, p. 94, doi: 10.1007/s11051-026-06621-1.

Azimzada, A., Tufenkji, N. and Wilkinson, K.J. (2017), "Transformations of silver nanoparticles in wastewater effluents: links to Ag bioavailability", *Environmental Science: Nano*, The Royal Society of Chemistry, Vol. 4 No. 6, pp. 1339–1349, doi: 10.1039/C7EN00093F.

Babu, S., Padmakumar, A., Akhila, K.S., Rubiyah, M.H., Varghese, S. and Thomas, B. (2025), "Life Cycle Assessment of Nanomaterials BT - Nanomaterial Green Synthesis", in Al-Khayri, J.M., Anju, T.R. and Jain, S.M. (Eds.), , Springer Nature Switzerland, Cham, pp. 387–424, doi: 10.1007/978-3-031-84643-4_13.

Benn, T.M. and Westerhoff, P. (2008), "Nanoparticle Silver Released into Water from Commercially Available Sock Fabrics", *Environmental Science & Technology*, American Chemical Society, Vol. 42 No. 11, pp. 4133–4139, doi: 10.1021/es7032718.

Bondarenko, O., Juganson, K., Ivask, A., Kasemets, K., Mortimer, M. and Kahru, A. (2013), "Toxicity of Ag, CuO and ZnO nanoparticles to selected environmentally relevant test organisms and mammalian cells in vitro: a critical review", *Archives of Toxicology*, Vol. 87 No. 7, pp. 1181–1200, doi: 10.1007/s00204-013-1079-4.

Bundschuh, M., Filser, J., Lüderwald, S., McKee, M.S., Metreveli, G., Schaumann, G.E., Schulz, R., *et al.* (2018), "Nanoparticles in the environment: where do we come from, where do we go to?", *Environmental Sciences Europe*, Vol. 30 No. 1, p. 6, doi: 10.1186/s12302-018-0132-6.

Cavazzoli, S., Ferrentino, R., Scopetani, C., Monperrus, M. and Andreottola, G. (2023), "Analysis of micro- and nanoplastics in wastewater treatment plants: key steps and environmental risk considerations.", *Environmental Monitoring and Assessment*, Netherlands, Vol. 195 No. 12, p. 1483, doi: 10.1007/s10661-023-12030-x.

Chakraborty, S. and Lynch, I. (2025), "Biomolecular Transformations Shape the Environmental Fate of Nanoscale and Emerging Materials.", *Accounts of Chemical Research*, United States, Vol. 58 No. 21, pp. 3316–3327, doi: 10.1021/acs.accounts.5c00587.

Chaudhari, N., Shinde, S., Havelikar, U., Desai, S. and Patel, V. (2025), "Recent approaches of toxicity-related attributes of NMs: A systematic review", *Nano Trends*, Vol. 12, p. 100160, doi: https://doi.org/10.1016/j.nwnano.2025.100160.

Chen, Y., Meng, Y., Liu, G., Huang, X. and Chai, G. (2024), "Probabilistic Estimation of Airborne

ACCEPTED MANUSCRIPT

Micro- and Nanoplastic Intake in Humans", *Environmental Science & Technology*, Vol. 58 No. 21, pp. 9071–9081, doi: 10.1021/acs.est.3c09189.

Cornelis, G., Hund-Rinke, K., Kuhlbusch, T., van den Brink, N. and Nickel, C. (2014), "Fate and Bioavailability of Engineered Nanoparticles in Soils: A Review", *Critical Reviews in Environmental Science and Technology*, Taylor & Francis, Vol. 44 No. 24, pp. 2720–2764, doi: 10.1080/10643389.2013.829767.

David, C.A., Galceran, J., Rey-Castro, C., Puy, J., Companys, E., Salvador, J., Monné, J., *et al.* (2012), "Dissolution Kinetics and Solubility of ZnO Nanoparticles Followed by AGNES", *The Journal of Physical Chemistry C*, American Chemical Society, Vol. 116 No. 21, pp. 11758–11767, doi: 10.1021/jp301671b.

Dimkpa, C.O. and Bindraban, P.S. (2018), "Nanofertilizers: New Products for the Industry?", *Journal of Agricultural and Food Chemistry*, American Chemical Society, Vol. 66 No. 26, pp. 6462–6473, doi: 10.1021/acs.jafc.7b02150.

Ellis (Bradford), L.-J.A., Njel, C., Chetwynd, A.J. and Lynch, I. (2026), "Understanding the surface chemistry of long term environmentally transformed nanomaterials for regulatory assessment", *Environmental Science: Nano*, Vol. 13 No. 6, pp. 2670–2684, doi: 10.1039/d6en00256k.

Feculak, M., Loureiro, S., White, J.C., Xing, B., Wu, K.C.-W., Sheteiwy, M.S., Gao, Y., *et al.* (2025), "Engineered nanoparticle transformations: Rethinking toxicity in water", *Nano Today*, Vol. 65, p. 102804, doi: https://doi.org/10.1016/j.nantod.2025.102804.

Froggett, S.J., Clancy, S.F., Boverhof, D.R. and Canady, R.A. (2014), "A review and perspective of existing research on the release of nanomaterials from solid nanocomposites", *Particle and Fibre Toxicology*, Vol. 11 No. 1, p. 17, doi: 10.1186/1743-8977-11-17.

Gawande, M.B., Goswami, A., Felpin, F.-X., Asefa, T., Huang, X., Silva, R., Zou, X., *et al.* (2016), "Cu and Cu-Based Nanoparticles: Synthesis and Applications in Catalysis", *Chemical Reviews*, American Chemical Society, Vol. 116 No. 6, pp. 3722–3811, doi: 10.1021/acs.chemrev.5b00482.

Gondikas, A.P., Kammer, F. von der, Reed, R.B., Wagner, S., Ranville, J.F. and Hofmann, T. (2014), "Release of TiO2 Nanoparticles from Sunscreens into Surface Waters: A One-Year Survey at the Old Danube Recreational Lake", *Environmental Science & Technology*, American Chemical Society, Vol. 48 No. 10, pp. 5415–5422, doi: 10.1021/es405596y.

Gottschalk, F., Sonderer, T., Scholz, R.W. and Nowack, B. (2009), "Modeled Environmental Concentrations of Engineered Nanomaterials (TiO2, ZnO, Ag, CNT, Fullerenes) for Different Regions", *Environmental Science & Technology*, American Chemical Society, Vol. 43 No. 24, pp. 9216–9222, doi: 10.1021/es9015553.

Gottschalk, F., Sun, T. and Nowack, B. (2013), "Environmental concentrations of engineered nanomaterials: Review of modeling and analytical studies", *Environmental Pollution*, Vol. 181, pp. 287–300, doi: https://doi.org/10.1016/j.envpol.2013.06.003.

Grillo, R., Rosa, A.H. and Fraceto, L.F. (2015), "Engineered nanoparticles and organic matter: A review of the state-of-the-art", *Chemosphere*, Vol. 119, pp. 608–619, doi: https://doi.org/10.1016/j.chemosphere.2014.07.049.

Hicks, A. (2025), "Life cycle assessment and engineered nanomaterials: looking to the past to inform the future – considering nanoscale silver as an example", *NanoImpact*, Vol. 39, p. 100569, doi: https://doi.org/10.1016/j.impact.2025.100569.

Hosny, S., Gaber, G.A., Ragab, M.S., Ragheb, M.A., Anter, M. and Mohamed, L.Z. (2026), "A Comprehensive Review of Silver Nanoparticles (AgNPs): Synthesis Strategies, Toxicity

ACCEPTED MANUSCRIPT

Concerns, Biomedical Applications, AI-Driven Advancements, Challenges, and Future Perspectives", *Arabian Journal for Science and Engineering*, Vol. 51 No. 11, pp. 13667–13714, doi: 10.1007/s13369-025-10612-0.

Hotze, E.M., Phenrat, T. and Lowry, G. V. (2010), "Nanoparticle Aggregation: Challenges to Understanding Transport and Reactivity in the Environment", *Journal of Environmental Quality*, John Wiley & Sons, Ltd, Vol. 39 No. 6, pp. 1909–1924, doi: https://doi.org/10.2134/jeq2009.0462.

Hou, W.-C., Westerhoff, P. and Posner, J.D. (2013), "Biological accumulation of engineered nanomaterials: a review of current knowledge", *Environmental Science: Processes & Impacts*, The Royal Society of Chemistry, Vol. 15 No. 1, pp. 103–122, doi: 10.1039/C2EM30686G.

Hunt, N., Kestens, V., Rasmussen, K., Badetti, E., Soeteman-Hernández, L.G., Oomen, A.G., Peijnenburg, W., *et al.* (2025), "Regulatory preparedness for multicomponent nanomaterials: Current state, gaps and challenges of REACH", *NanoImpact*, Vol. 37, p. 100538, doi: https://doi.org/10.1016/j.impact.2024.100538.

Islam, S. (2025), "Toxicity and transport of nanoparticles in agriculture: effects of size, coating, and aging", *Frontiers in Nanotechnology*, Vol. Volume 7-2025, doi: 10.3389/fnano.2025.1622228.

Ivask, A., ElBadawy, A., Kaweeteerawat, C., Boren, D., Fischer, H., Ji, Z., Chang, C.H., *et al.* (2014), "Toxicity Mechanisms in Escherichia coli Vary for Silver Nanoparticles and Differ from Ionic Silver", *ACS Nano*, American Chemical Society, Vol. 8 No. 1, pp. 374–386, doi: 10.1021/nn4044047.

Joudeh, N. and Linke, D. (2022), "Nanoparticle classification, physicochemical properties, characterization, and applications: a comprehensive review for biologists.", *Journal of Nanobiotechnology*, England, Vol. 20 No. 1, p. 262, doi: 10.1186/s12951-022-01477-8.

Judy, J.D., Unrine, J.M. and Bertsch, P.M. (2011), "Evidence for Biomagnification of Gold Nanoparticles within a Terrestrial Food Chain", *Environmental Science & Technology*, American Chemical Society, Vol. 45 No. 2, pp. 776–781, doi: 10.1021/es103031a.

Kah, M., Kookana, R.S., Gogos, A. and Bucheli, T.D. (2018), "A critical evaluation of nanopesticides and nanofertilizers against their conventional analogues", *Nature Nanotechnology*, Vol. 13 No. 8, pp. 677–684, doi: 10.1038/s41565-018-0131-1.

von der Kammer, F., Ferguson, P.L., Holden, P.A., Masion, A., Rogers, K.R., Klaine, S.J., Koelmans, A.A., *et al.* (2012), "Analysis of engineered nanomaterials in complex matrices (environment and biota): General considerations and conceptual case studies*", *Environmental Toxicology and Chemistry*, Vol. 31 No. 1, pp. 32–49, doi: 10.1002/etc.723.

Keller, A.A., McFerran, S., Lazareva, A. and Suh, S. (2013), "Global life cycle releases of engineered nanomaterials", *Journal of Nanoparticle Research*, Vol. 15 No. 6, p. 1692, doi: 10.1007/s11051-013-1692-4.

Keller, A.A., Wang, H., Zhou, D., Lenihan, H.S., Cherr, G., Cardinale, B.J., Miller, R., *et al.* (2010), "Stability and aggregation of metal oxide nanoparticles in natural aqueous matrices.", *Environmental Science & Technology*, United States, Vol. 44 No. 6, pp. 1962–1967, doi: 10.1021/es902987d.

Kiser, M.A., Westerhoff, P., Benn, T., Wang, Y., Pérez-Rivera, J. and Hristovski, K. (2009), "Titanium Nanomaterial Removal and Release from Wastewater Treatment Plants", *Environmental Science & Technology*, American Chemical Society, Vol. 43 No. 17, pp. 6757–6763, doi: 10.1021/es901102n.

Kumar, A., Soratur, A., Kumar, S., Sarkar, A., Krishna, S.S.S., Kiruba-Sankar, R., Jha, D.K., *et al.*

ACCEPTED MANUSCRIPT

(2026), “Organ-specific bioaccumulation of trace elements in commercial marine finfish from the South Andaman Islands and associated human health risk assessment”, *Discover Oceans*, Vol. 3 No. 1, p. 35, doi: 10.1007/s44289-026-00150-4.

Labille, J., Feng, J., Botta, C., Borschneck, D., Sammut, M., Cabie, M., Auffan, M., *et al.* (2010), “Aging of TiO2 nanocomposites used in sunscreen. Dispersion and fate of the degradation products in aqueous environment”, *Environmental Pollution*, Vol. 158 No. 12, pp. 3482–3489, doi: https://doi.org/10.1016/j.envpol.2010.02.012.

Laborda, F., Jiménez-Lamana, J., Bolea, E. and Castillo, J.R. (2013), “Critical considerations for the determination of nanoparticle number concentrations, size and number size distributions by single particle ICP-MS”, *Journal of Analytical Atomic Spectrometry*, The Royal Society of Chemistry, Vol. 28 No. 8, pp. 1220–1232, doi: 10.1039/C3JA50100K.

Le, S., Nguyen, B., Bui, H., Lê, N., Dinh, H., Le, K., Cao Truong, S., *et al.* (2025), “Bioaccumulation of Selected Heavy Metals in Bivalve Molluscs from Northeastern Vietnam: Implications for Safe Seafood Utilization”, *Research in Ecology*, doi: 10.30564/re.v7i3.9805.

Lee, M., Jung, S., Yoon, C. and Hwang, S.H. (2026), “Distribution and Concentration of Airborne Per- and Polyfluoroalkyl Substances (PFAS) in Indoor and Outdoor Environments: A Systematic Review”, *Safety and Health at Work*, Vol. 17 No. 1, pp. 12–23, doi: https://doi.org/10.1016/j.shaw.2026.01.002.

Levard, C., Hotze, E.M., Colman, B.P., Dale, A.L., Truong, L., Yang, X.Y., Bone, A.J., *et al.* (2013), “Sulfidation of Silver Nanoparticles: Natural Antidote to Their Toxicity”, *Environmental Science & Technology*, American Chemical Society, Vol. 47 No. 23, pp. 13440–13448, doi: 10.1021/es403527n.

Li, M., Lu, J., Chen, Z. and Amine, K. (2018), “30 Years of Lithium-Ion Batteries”, *Advanced Materials*, Vol. 30, doi: 10.1002/adma.201800561.

Liu, J. and Hurt, R.H. (2010), “Ion Release Kinetics and Particle Persistence in Aqueous Nano-Silver Colloids”, *Environmental Science & Technology*, American Chemical Society, Vol. 44 No. 6, pp. 2169–2175, doi: 10.1021/es9035557.

Lowry, G. V, Gregory, K.B., Apte, S.C. and Lead, J.R. (2012), “Transformations of Nanomaterials in the Environment”, *Environmental Science & Technology*, American Chemical Society, Vol. 46 No. 13, pp. 6893–6899, doi: 10.1021/es300839e.

Martínez, G., Merinero, M., Pérez-Aranda, M., Pérez-Soriano, E.M., Ortiz, T., Begines, B. and Alcudia, A. (2020), “Environmental Impact of Nanoparticles’ Application as an Emerging Technology: A Review.”, *Materials (Basel, Switzerland)*, Switzerland, Vol. 14 No. 1, doi: 10.3390/ma14010166.

Maynard, A.D. (2015), “Navigating the fourth industrial revolution”, *Nature Nanotechnology*, Vol. 10 No. 12, pp. 1005–1006, doi: 10.1038/nnano.2015.286.

Meng, Z., Zheng, L., Fang, H., Yang, P., Wang, B., Li, L., Wang, M., *et al.* (2023), “Single Particle Inductively Coupled Plasma Time-of-Flight Mass Spectrometry—A Powerful Tool for the Analysis of Nanoparticles in the Environment”, *Processes*, doi: 10.3390/pr11041237.

Miller, R.J., Lenihan, H.S., Muller, E.B., Tseng, N., Hanna, S.K. and Keller, A.A. (2010), “Impacts of Metal Oxide Nanoparticles on Marine Phytoplankton”, *Environmental Science & Technology*, American Chemical Society, Vol. 44 No. 19, pp. 7329–7334, doi: 10.1021/es100247x.

Mitchell, M.J., Billingsley, M.M., Haley, R.M., Wechsler, M.E., Peppas, N.A. and Langer, R. (2021), “Engineering precision nanoparticles for drug delivery”, *Nature Reviews Drug*

ACCEPTED MANUSCRIPT

Discovery, Vol. 20 No. 2, pp. 101–124, doi: 10.1038/s41573-020-0090-8.

Montaño, M.D., Olesik, J.W., Barber, A.G., Challis, K. and Ranville, J.F. (2016), "Single Particle ICP-MS: Advances toward routine analysis of nanomaterials", *Analytical and Bioanalytical Chemistry*, Vol. 408 No. 19, pp. 5053–5074, doi: 10.1007/s00216-016-9676-8.

Nel, A., Xia, T., Mädler, L. and Li, N. (2006), "Toxic Potential of Materials at the Nanolevel", *Science*, American Association for the Advancement of Science, Vol. 311 No. 5761, pp. 622–627, doi: 10.1126/science.1114397.

Nel, A.E., Nasser, E., Godwin, H., Avery, D., Bahadori, T., Bergeson, L., Beryt, E., *et al.* (2013), "A Multi-Stakeholder Perspective on the Use of Alternative Test Strategies for Nanomaterial Safety Assessment", *ACS Nano*, American Chemical Society, Vol. 7 No. 8, pp. 6422–6433, doi: 10.1021/nn4037927.

Nowack, B., Ranville, J.F., Diamond, S., Gallego-Urrea, J.A., Metcalfe, C., Rose, J., Horne, N., *et al.* (2012), "Potential scenarios for nanomaterial release and subsequent alteration in the environment*", *Environmental Toxicology and Chemistry*, Vol. 31 No. 1, pp. 50–59, doi: 10.1002/etc.726.

Oladipo, A.A. (2026), "The double-edged nanoparticle: remediation benefits vs. mechanistic toxicity risks in aquatic systems", *Environmental Science: Nano*, Vol. 13 No. 1, pp. 79–105, doi: 10.1039/d5en00831j.

Oros, A. (2025), "Bioaccumulation and Trophic Transfer of Heavy Metals in Marine Fish: Ecological and Ecosystem-Level Impacts", *Journal of Xenobiotics*, doi: 10.3390/jox15020059.

Padmanabhan, S., Alsayaydeh, J.A.J., Kumar, T.V., Ganesan, S., Bacarra, R. and Mohamood, N. Bin. (2026), "A comprehensive review of nanocatalysts for sustainable plastic waste valorization in circular economy", *Discover Sustainability*, doi: 10.1007/s43621-026-03902-6.

Pérez-López, A., Cotillas, S., Santos, A. and Domínguez, C.M. (2026), "Micro- and nanoplastics removal from water and solid matrices: Technologies, challenges, and future perspectives", *Environmental Research*, Vol. 299, p. 124295, doi: https://doi.org/10.1016/j.envres.2026.124295.

Petersen, E.J., Diamond, S.A., Kennedy, A.J., Goss, G.G., Ho, K., Lead, J., Hanna, S.K., *et al.* (2015), "Adapting OECD Aquatic Toxicity Tests for Use with Manufactured Nanomaterials: Key Issues and Consensus Recommendations", *Environmental Science & Technology*, American Chemical Society, Vol. 49 No. 16, pp. 9532–9547, doi: 10.1021/acs.est.5b00997.

Piccinno, F., Gottschalk, F., Seeger, S. and Nowack, B. (2012), "Industrial production quantities and uses of ten engineered nanomaterials in Europe and the world", *Journal of Nanoparticle Research*, Vol. 14 No. 9, p. 1109, doi: 10.1007/s11051-012-1109-9.

Quik, J.T.K., Stuart, M.C., Wouterse, M., Peijnenburg, W., Hendriks, A.J. and van de Meent, D. (2012), "Natural colloids are the dominant factor in the sedimentation of nanoparticles", *Environmental Toxicology and Chemistry*, Vol. 31 No. 5, pp. 1019–1022, doi: 10.1002/etc.1783.

Ramadan, B.S., Wibowo, Y.G., Anwar, D. and Maryani, A.T. (2024), "A Review of Life Cycle Assessment of Nanomaterials-Based Adsorbent for Environmental Remediation", Vol. 26 No. 7.

Ridoy, M.N. and Supto, S.T.J. (2025), "Environmental Impacts and Sustainability of Nanomaterials in Water and Soil Systems", *Materials Proceedings*, doi: 10.3390/materproc2025026006.

Stankus, D.P., Lohse, S.E., Hutchison, J.E. and Nason, J.A. (2011), "Interactions between Natural

ACCEPTED MANUSCRIPT

Organic Matter and Gold Nanoparticles Stabilized with Different Organic Capping Agents", *Environmental Science & Technology*, American Chemical Society, Vol. 45 No. 8, pp. 3238–3244, doi: 10.1021/es102603p.

Sun, T.Y., Bornhöft, N.A., Hungerbühler, K. and Nowack, B. (2016), "Dynamic Probabilistic Modeling of Environmental Emissions of Engineered Nanomaterials", *Environmental Science & Technology*, American Chemical Society, Vol. 50 No. 9, pp. 4701–4711, doi: 10.1021/acs.est.5b05828.

Vance, M.E., Kuiken, T., Vejerano, E.P., McGinnis, S.P., Hochella Jr., M.F., Rejeski, D. and Hull, M.S. (2015), "Nanotechnology in the real world: Redeveloping the nanomaterial consumer products inventory", *Beilstein Journal of Nanotechnology*, Beilstein-Institut, Vol. 6, pp. 1769–1780, doi: 10.3762/bjnano.6.181.

Venkatesan, J., Lee, J.-Y., Kang, D.S., Anil, S., Kim, S.-K., Shim, M.S. and Kim, D.G. (2017), "Antimicrobial and anticancer activities of porous chitosan-alginate biosynthesized silver nanoparticles", *International Journal of Biological Macromolecules*, Vol. 98, pp. 515–525, doi: https://doi.org/10.1016/j.ijbiomac.2017.01.120.

Villarruel, C., Buurma, E.E., Ranville, J. and Fairbrother, D.H. (2026), "Probing the Environmental Persistence of Inorganic Nanoparticles and Micro(nano)plastics by Single-Particle ICP-MS", *Environmental Science & Technology*, Vol. 60 No. 14, pp. 10435–10446, doi: 10.1021/acs.est.5c18660.

Westerhoff, P., Song, G., Hristovski, K. and Kiser, M.A. (2011), "Occurrence and removal of titanium at full scale wastewater treatment plants: implications for TiO2 nanomaterials", *Journal of Environmental Monitoring*, The Royal Society of Chemistry, Vol. 13 No. 5, pp. 1195–1203, doi: 10.1039/C1EM10017C.

Whitley, A.R., Levard, C., Oostveen, E., Bertsch, P.M., Matocha, C.J., Kammer, F. von der and Unrine, J.M. (2013), "Behavior of Ag nanoparticles in soil: Effects of particle surface coating, aging and sewage sludge amendment", *Environmental Pollution*, Vol. 182, pp. 141–149, doi: https://doi.org/10.1016/j.envpol.2013.06.027.

ACCEPTED MANUSCRIPT