

Critical Review

Nanomaterials in the Environment: Behavior, Fate, Bioavailability, and Effects—An Updated Review

Jamie R. Lead,^{a,*} Graeme E. Batley,^{b,*} Pedro J. J. Alvarez,^c Marie-Noëlle Croteau,^d Richard D. Handy,^e Michael J. McLaughlin,^f Jonathan D. Judy,^g and Kristin Schirmer^{h,i,j}

^aCenter for Environmental Nanoscience and Risk, Department of Environmental Health Sciences, Arnold School of Public Health, University of South Carolina, Columbia, South Carolina, USA

^bCentre for Environmental Contaminants Research, CSIRO Land and Water, Kirrawee, New South Wales, Australia

^cDepartment of Civil and Environmental Engineering, Rice University, Houston, Texas, USA

^dUS Geological Survey, Menlo Park, California, USA

^eUniversity of Plymouth, Plymouth, Devon, United Kingdom

^fUniversity of Adelaide, Glen Osmond, South Australia, Australia

^gSoil and Water Sciences Department, University of Florida, Gainesville, Florida, USA

^hDepartment of Environmental Toxicology, Swiss Federal Institute of Aquatic Science and Technology, Eawag, Dübendorf, Switzerland

ⁱSchool of Architecture, Civil and Environmental Engineering, Federal Institute of Technology Lausanne, Lausanne, Switzerland

^jInstitute of Biogeochemistry and Pollutant Dynamics, Swiss Federal Institute of Technology Zürich, Zürich, Switzerland

Abstract: The present review covers developments in studies of nanomaterials (NMs) in the environment since our much cited review in 2008. We discuss novel insights into fate and behavior, metrology, transformations, bioavailability, toxicity mechanisms, and environmental impacts, with a focus on terrestrial and aquatic systems. Overall, the findings were that: 1) despite substantial developments, critical gaps remain, in large part due to the lack of analytical, modeling, and field capabilities, and also due to the breadth and complexity of the area; 2) a key knowledge gap is the lack of data on environmental concentrations and dosimetry generally; 3) substantial evidence shows that there are nanospecific effects (different from the effects of both ions and larger particles) on the environment in terms of fate, bioavailability, and toxicity, but this is not consistent for all NMs, species, and relevant processes; 4) a paradigm is emerging that NMs are less toxic than equivalent dissolved materials but more toxic than the corresponding bulk materials; and 5) translation of incompletely understood science into regulation and policy continues to be challenging. There is a developing consensus that NMs may pose a relatively low environmental risk, but because of uncertainty and lack of data in many areas, definitive conclusions cannot be drawn. In addition, this emerging consensus will likely change rapidly with qualitative changes in the technology and increased future discharges. *Environ Toxicol Chem* 2018;37:2029–2063. © 2018 The Authors. *Environmental Toxicology and Chemistry* published by Wiley Periodicals, Inc. on behalf of SETAC.

Keywords: Nanomaterials; Nanoecotoxicity; Hazard/risk assessment; Nanometrology; Aquatic and soil organisms

This article includes online-only Supplemental Data.

¹These authors contributed equally to this manuscript.

* Address correspondence to JLEAD@mailbox.sc.edu and Graeme.batley@csiro.au

This is an open access article under the terms of the Creative Commons Attribution-NonCommercial-NoDerivs License, which permits use and distribution in any medium, provided the original work is properly cited, the use is non-commercial and no modifications or adaptations are made.

Published online 6 April 2018 in Wiley Online Library

(wileyonlinelibrary.com).

DOI: 10.1002/etc.4147

A PIONEERING REVIEW

In 2007, Steve Klaine foresaw the need for a paper to summarize progress in the burgeoning field of nanomaterials in the environment. With the aid of 8 other researchers from 3 countries, his pioneering review was published in *Environmental Toxicology and Chemistry (ET&C)* in 2008 (Klaine et al. 2008), going on to become the most highly cited paper ever in *ET&C*, and possibly the most cited paper on nanoparticles in the environment and nanotoxicology. Steve put the review together

almost at the start of this area as a separate discipline, when there was growing recognition of the importance of nanoparticles and their potential risk, a lot of questions, and fairly limited data and answers. In late 2015, Steve began a conversation about updating that paper before ill health overtook him. The paper "In Memoriam: Stephen J. Klaine" by Rice et al. (2016) gives a flavor of Steve's importance to family and friends, to this journal, and to the discipline. In his absence, the original coauthors, with other colleagues, took over the production of the present review. We very much hope it is a fitting tribute to Steve Klaine.

INTRODUCTION

In 2008, we produced one of the first comprehensive reviews of nanomaterials (NMs) in the environment (Klaine et al. 2008), which has since been very highly cited. The pace of NM research, development, and application evident at that time has continued and increased. Rapid increases in citations and patents granted are evident. The widely cited Nanotechnology Consumer Products Inventory showed that nanoenabled products increased in number from 50 to nearly 2000 (Vance et al. 2015) and continue to grow. Despite the limitations of the Inventory database, it provides evidence of substantial increases in nanotechnology use. In addition, there is a large and growing body of nanoenvironmental and nanoecotoxicological research (Selck et al. 2016). This research is driven by questions of environmental risk because of the predicted rapid increases in environmental concentrations (Gottschalk et al. 2013), the known bioavailability (Luoma et al. 2014) and deleterious biological effects (Fabrega et al. 2013), the suspected novel behavior of some NMs in environmental and biological systems (Luoma et al. 2014; Taylor et al. 2016), and the consequent complexities of risk assessment (Laux et al. 2018). In addition, modeling projections using a "business as usual" scenario suggest that production will more than triple by 2020, with much of the output eventually being discharged to the environment (Sun et al. 2017). It is therefore timely to review progress over the past decade. In particular, we will consider the advances that have been achieved, where the most critical gaps and opportunities are, and where improvements in our knowledge base are needed.

With the increasing use of NMs from early 2000, the question of whether they pose a risk to the environment has loomed large. In 2008, there was a lack of basic data on fate and behavior, ecotoxicity, and related issues (Klaine et al. 2008). The field has provided a great deal of high-quality advances in all of these areas, but substantial uncertainties and data gaps remain. Regulatory agencies needed (and still need) to decide whether NMs as a class must be managed differently from larger micron-sized particles or the dissolved phase (Pettitt and Lead 2013; Hund-Rinke et al. 2016). This relevant question can be posed as follows: does the nanoscale imply a greater risk to aquatic and terrestrial ecosystem health through novel behavior related to extra reactivity, increased transport, and interaction with subcellular components, leading to more

serious adverse interactions with organisms? The novel properties necessary for technology and innovation, such as the spatial constraint of electronic properties and high specific surface area might, but do not necessarily, imply novel environmental and biological behavior. For example, we now know that the critical effector of the antibacterial activity of Ag NMs is the released Ag ions rather than the NMs themselves (Xiu et al. 2011), at least for single-species cultures. Accordingly, any effect of Ag NM size, coating, or shape on single-species bacteria is indirect, because these factors mainly influence transport, bioavailability, and the rate and extent of Ag⁺ release (Xiu et al. 2011; for details, see the discussion in the Supplemental Data). For biofilms and eukaryotes, the situation is not as clear cut, and there is likely to be a role for the NMs, separate from the ions. Many studies show significant size-dependent effects, which are not readily explained by metal ion dissolution, the speciation chemistry of the solutes released, or their chemical reactivity. NMs show a separate bioavailable fraction and biological effects (van Hoecke et al. 2009; Croteau et al. 2011a, 2011b; Yin et al. 2011; Pan et al. 2012; Al-Jubory and Handy 2013; Pang et al. 2013; Cozzari et al. 2015; Taylor et al. 2016). For instance, Taylor et al. (2016) showed that tightly constrained (in size and oxidation state) and well-characterized ceria NMs gave different algal transcriptomic and metabolomic responses compared with larger scale particles and dissolved phase Ce, although in all cases toxicity was low. Confounding factors to such studies lie in: 1) ensuring that NMs and larger particles are only different with respect to one variable, which is difficult when physical and chemical properties are size dependent, as is the case for ceria and oxidation state; 2) rapid transformations of NMs occurring in complex media such as environmental and exposure media; and 3) precipitation of the dissolved controls after exposure (Merrifield RC et al. 2013). In addition, the limited knowledge of hazard and exposure (Klaine et al. 2012) creates large uncertainties in understanding risk. The details of potential nanoeffects are given in later sections.

Issues such as nanometrology, dosimetry, and transformations, especially in complex media, which were previously just coming to the attention of the scientific community, have now been brought more sharply into focus (Baalousha et al. 2012; Lowry et al. 2012a; Selck et al. 2016). Transformations may reduce toxicity (Levard et al. 2012), although this is not always the case (Wang Q et al. 2013). Questions remain about the timescales and reversibility of transformations, and their effects on persistence and bioaccumulation. In addition, the many new variants of NMs entering the market (with their system-specific transformation products often unknown) hinder the accurate modeling and prediction of NM fate and impacts.

The present review aims to examine the new knowledge of NMs gained since publication of the Klaine et al. (2008) review, particularly in aquatic and terrestrial systems, and to discuss new questions arising from research conducted since 2008 in terms of the environmental fate and effects of NMs.

SELECTED RECENT ADVANCES IN NM COMPOSITION AND METROLOGY

Surface modification of NMs

The study of coatings to modify surface properties and alter aggregation behavior in the environment has advanced significantly since our last review (Klaine et al. 2008; for instance, see Tolaymat et al. 2010). A number of studies have investigated the effects of citrate, polyvinylpyrrolidone (PVP), polyethylene glycol, and other coatings for stabilizing NMs (Angel et al. 2013). Some, but not all, of these coatings are known to have associated biological activity. Natural organic matter (NOM; particularly humic substances) has been shown to perform a similar function, with a range of NMs affecting both fate and biological effects (Yin et al. 2015), although often with concomitant effects on the core material (Merrifield et al. 2017a). The use of coatings on carbon-based materials is rarer, with changing properties commonly achieved via surface functionalization (Balasubramanian and Burghard 2005). Questions related to the changing nature of surface chemistry (e.g., the formation of ecocoronas and protein coronas) are being addressed (Manciualea et al. 2009; Mudunkotuwa and Grassian 2015) but require further work. There are limited or no data on the kinetics and extent of ecocorona and protein corona exchange with synthetic coatings, for instance.

New NMs

Although Ag, zinc oxide (ZnO), copper oxide (CuO), cerium dioxide (CeO₂), titanium dioxide (TiO₂), iron oxide (FeO), fullerenes, carbon nanotubes (CNTs), and a small number of others remain the most widely used and researched NMs, some newer NMs have been produced in recent years. (Note: throughout, we use symbols as shorthand and do not make assumptions about stoichiometry—e.g., FeO stands in for a range of Fe(II) and (III) oxides). The greatest interest and development has been in broad classes of materials including nanocomposites and nanohybrids, sometimes referred to as either multiple-component nanohybrids (Wang R et al. 2015), core-shell nanocrystals (Chaudhuri and Paria 2012), or oligomer NMs (Miao et al. 2016). The terminology is confusing, but one distinction is that nanocomposites are NMs enclosed or encapsulated with other materials, which may not have nanoscale features, whereas nanohybrids are the linking of 2 or more discrete NMs to give different functionality (Saleh et al. 2015). Such NMs can be based on semiconductor substrates such as GaAs, CdSe, CdS, SiGe, and others, modified with shells and coatings, along with mixtures of carbon-carbon and carbon-metal or metal oxides (Wu et al. 2015). Quantum dots are one such hybrid and have been in use for several decades. Their applications are widespread and include medical diagnostics, solar cells, nanoelectronics, detectors, and photocatalysts. For many composites and hybrids, however, their current use is relatively small in many sectors, and their present likely impact on the environment is low, except in the hypothetical case of a major spill. Nevertheless, use is likely to increase rapidly, and they are therefore likely to present a

significant concern in the near future. Future research will need to focus on the release rates, persistence, bioavailability, and toxicity of these nanohybrids.

Klaine et al. (2008) called for the development of new standards of various sorts. At around the same time, there was much discussion of test materials, reference materials, and certified reference materials between producers of such materials and so-called consumers (toxicologists, ecotoxicologists, and others). Relatively simple test and reference materials of good quality are now available from bodies such as the National Institute of Standards and Technology in the United States, the European Commission's Joint Research Centre, and others, including commercial sources, in North America, Europe, and elsewhere. Clearly, study aims drive the choice of such materials, and researchers must choose the correct parameters by which the materials are referenced (size, concentration, etc.).

Nanometrology

Characterization and metrology have recently been reviewed in detail by Baalousha and Lead (2015), and detailed discussions about technical advances can be found there. Improved metrology is required to refine our understanding of environmental concentrations and the forms of NMs, as well as to enable more accurate dosimetry in ecotoxicology testing, which would define the dose-response relationship more exactly. Current methods are limited for various reasons, including inadequate sensitivity and selectivity in relation to the complexity of both NMs and environmental conditions, lack of resolution (for imaging techniques), an inability to provide full quantification, and a lack of broad applicability. For instance, ultraviolet (UV)-visible spectroscopy is highly selective for metals such as Ag, Au, and Cu in relatively simple systems (Zook et al. 2011), but produces lower quality information in more complex systems and is not very sensitive. Sensitivity constraints can be partially alleviated by using longer path lengths for transmission through the sample. Modern transmission electron microscopy (TEM) systems have sub-nm resolution and can perform single-particle analysis for a wide variety of properties (size, composition, shape, chemical speciation). Although not widely available, in-liquid imaging can be performed (Zeng et al. 2017), sample handling methods can overcome drying and beam damage issues (Prasad et al. 2015), and atomic force microscopy can be used to analyze liquid samples (Lead et al. 2005). Dynamic light scattering is frequently used for size measurement, and again it works very well for simple samples that are monodispersed, but size data are often inaccurate in aggregating/aggregated systems for a variety of reasons (Liu et al. 2012).

In the years since 2008, the situation has been improved by the development of new methods such as NP tracking analysis, hyphenated methods such as field flow fractionation with inductively coupled plasma-mass spectrometry (FFF-ICP-MS; Meisterjahn et al. 2016), and especially single-particle (SP)-ICP-MS (Reed et al. 2012; Bi et al. 2014; Yang et al. 2016; Tharaud et al. 2017). Nanoparticle tracking analysis may give more accurate data compared to dynamic light scattering

(Domingos et al. 2009), and SP-ICP-MS allows the single and ensemble analysis of particle number and concentration on an ion-specific basis (Merrifield et al. 2017a). More recently, SC-ICP-MS (Merrifield et al. 2018) has allowed the quantification of cellularly internalized concentrations of NMs, and these ICP-based developments are proving to be extremely powerful. The hyphenated methods such as FFF-ICP-MS and size exclusion chromatography-ICP-MS, which have been widely used (Meermann 2015), could also be combined to provide extra information. A limitation of the SP-ICP-MS method is the smallest size of NM that can be quantified (Lee et al. 2014), which is routinely larger than 20 to 40 nm and is element dependent. Various current developments such as very sensitive detectors, removal of ions during sample introduction, desolvation of the sample prior to the plasma, and use of isotopically pure materials should reduce the detectable NM size to nearly 1 nm in the near future. The SP-ICP-MS technique has been adopted for a range of environmental studies (Pace et al. 2011, 2012; Hadioui et al. 2015), and its importance in validating exposure modeling has been discussed (Nowack et al. 2015). It can be clearly seen that all methods have advantages and limitations, and that no single metrological method will give all the data required. A multimethod approach (Domingos et al. 2009) has been developed and widely discussed that brings to bear a wide range of techniques for sample analysis, allowing more detailed, complete, accurate, and unbiased production of data on NM physicochemistry. This multimethod approach, employed on pristine and potentially transformed NMs, is recommended.

The reactivity and toxicity of NMs is largely dependent on their physical and chemical properties such as size, shape, specific surface area, elemental composition, surface functionalization and modification, crystalline structure, and other factors (Derfus et al. 2004; Lesniak et al. 2005). Depending on exposure conditions, such properties can affect the propensity of NMs to aggregate, attach to surfaces, release toxic metal ions, become passivated, or interact with various environmental or biological constituents such as humic substances, (muco)polysaccharides, and cellular debris, in a manner that affects their bioavailability, uptake, and toxicity (Liu et al. 2016, 2013; Sayes 2014), and so the need for metrology is clear.

As an example of why metrology is needed, the relationship between NM surface properties, such as surface energy and reactivity, and toxicity has received little attention. Some research shows that, despite faster dissolution of toxic metals such as Cd^{2+} and higher affinity for proteins, CdS NMs with higher surface energy are less toxic than lower energy NMs. This unexpected difference is possibly due to the greater propensity of NMs to adsorb to or react with biological protective barriers and/or background constituents that alter behavior and effects (Liu et al. 2016). Such reactions passivate their reactivity and decrease their bioavailability and toxicity. The need for appropriate metrology and linking to behavior and effects is clear.

Further discussion on nanometrology and its advances is provided in the Supplemental Data and in cited references.

ENVIRONMENTAL FATE AND BEHAVIOR OF NMs

The environmental fate and behavior of NMs is fundamental to understanding their effects on environmental health and is a key aspect of environmental risk assessment (Stone et al. 2010; Hartmann et al. 2014). Knowledge of the sources, pathways, transformations, and sinks for NMs will provide information on the compartments (water, sediment, and biota) that are exposed to the NMs. Furthermore, understanding the behavior of the NMs in the environment will allow specific hazards to be identified, such as the bioaccumulation potential in aquatic food webs (Luoma et al. 2014; Selck et al. 2016).

Physicochemical properties

The role of NM characteristics in environmental risk is widely appreciated (Hassellöv and Kaegi 2009), although the methods for quantifying these properties are not always appropriate or deployed appropriately. In our earlier review (Klaine et al. 2008), the role of transformations was mentioned briefly. However, physicochemical properties such as size, composition, surface energy, and specific surface area of the original, pristine (as-made or as-used) NMs are still important for 2 reasons. First, for risk assessment and management purposes, along with an assessment of NP risks and benefits, an understanding of the original (and transformed) NMs is required. Second, transformation behaviors and rates are critically dependent on these original properties. However, it is now known that the properties of the transformation products are far more significant than originally thought (Lowry et al. 2012a), and these are discussed in detail in the following sections.

Processes and transformations affecting NM fate and behavior in surface waters

Transformations of NMs are analogous to the problem of changes in speciation in metal fate, behavior, bioavailability, and effects. In 2008 we were only beginning to think about these issues, which have since been reviewed (Lowry et al. 2012a; Hartmann et al. 2014). Dramatic improvements in our knowledge have taken place, and the importance of transformations in complex media such as the environment is now better understood. Transformations can be subdivided into physical, chemical, and biological processes. Physical processes include aggregation, agglomeration, sedimentation, and deposition (in porous media). Chemical processes include dissolution and subsequent speciation changes, redox reactions (oxidation and sulfidation), photochemical reactions, and corona formation. Biologically mediated processes include biodegradation and biomodification, most likely microbially mediated. These are encapsulated in a conceptual process model (Figure 1). Although it is clear that transformations will depend on the nature of the NMs and the environmental conditions, the complexity and variability of both these factors make understanding and prediction extremely challenging.

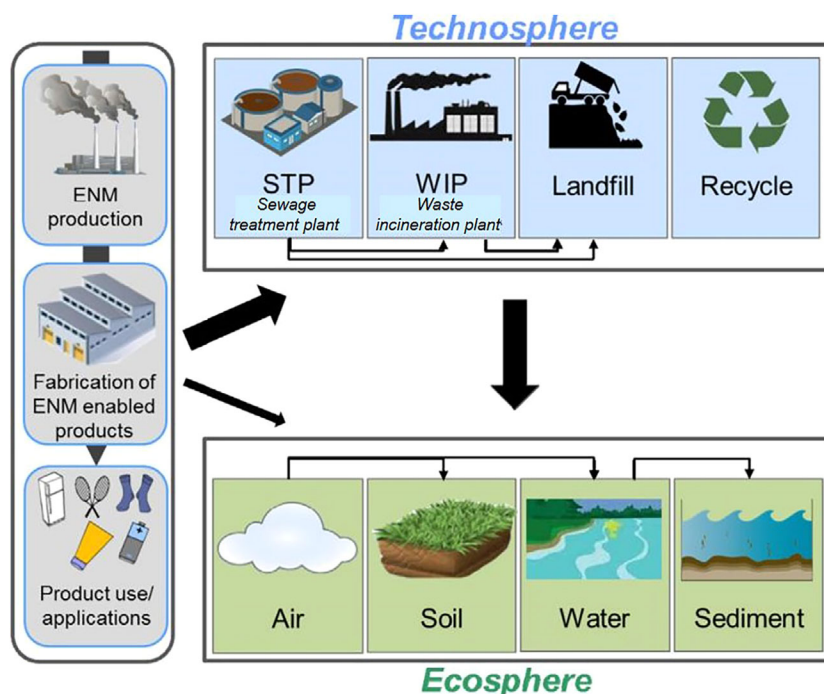


FIGURE 1: General structure of the material-flow model. The model's principle is to track engineered NM flows throughout the entire life cycle: engineered NM production; incorporation into products; engineered NM release from products during use; transport and fate of engineered NM between and within sewage treatment plants, waste incineration plants, landfill, and recycling processes (technosphere); transfer from technosphere to air, soil, water, and sediments (ecosphere); and transport within environmental compartments. The amounts of engineered NM in the compartments provide the basis for calculating the overall environmental concentrations of engineered NM. NM = nanomaterial; ROS = reactive oxygen species; NOM = natural organic matter; NP = nanoparticle. (From Sun et al. 2014.)

Dissolution and solubility

For some NMs such as ZnO, whose toxicity has been largely attributed to the ions (Franklin et al. 2007), solubility has a major influence on fate and toxicity. For other NMs such as Ag, CuO, and some quantum dots, intermediate dissolution and solubility suggest a role for both ions and particles (Hartmann et al. 2014; Leclerc and Wilkinson 2014), possibly with the ions having the direct biological impact and the NMs increasing both transport to the cell and local ionic concentrations. For carbon-based NMs and many inorganic NMs such as ceria and titania, whose solubility is low, dissolution and solubility become less important. Nevertheless, dissolution in microenvironments such as cellular vacuoles, where pH is reduced, may be important even for these NMs.

For the purpose of the present review and for nanoscience in the environment, the importance of the ion relative to the particle should be judged against the relative behavior and effects over relevant timescales. In toxicology, for instance, the importance of dissolution can be judged over the exposure time period, and this may be different from the equilibrium solubility. The solubility of nano-CeO₂ is extremely low, making it a useful material for studying NM effects directly without the complication of accounting for ionic behavior. For instance, nano-CeO₂ can be used to investigate the effects on behavior and toxicity of nano- compared with micron-sized materials (Rogers et al. 2010).

In addition to composition, several studies have shown that certain intrinsic NM properties affect dissolution and solubility,

including size (Tsiola et al. 2017), coating (Toncelli et al. 2017), and doping (Adeleye et al. 2018). A major issue in this area, as in other areas where attempts are made to correlate properties and behavior, has been the difficulty of varying single properties of NMs while leaving other confounding factors unchanged, although this is being done more successfully now than in 2008 (see Adeleye et al. 2018). In addition, studies have investigated the impacts of external factors such as NOM on NM behavior and have concluded that dissolution may be enhanced (Xiao et al. 2018), along with particle ripening and precipitation of new NMs (Merrifield et al. 2017b). Organic materials can also significantly impact NM dissolution (Luoma et al. 2016), but their influence is complex, and differing study data have not been fully rationalized (Aiken et al. 2011). Natural waters from a variety of systems have been studied (Odzak et al. 2015, 2017), and these complex systems have been shown to substantially affect dissolution (Wasmuth et al. 2016).

Aggregation, sedimentation, and deposition

The interaction between charged colloidal particles in solution has been described by the Derjaguin, Landau, Verwey, and Overbeek (DLVO) theory as the combination of repulsive and attractive forces. The application of this theory to NMs has been somewhat confounded by their polydispersity, complexity, and nonsphericity (Hotze et al. 2010), although in general the theory works well (Aich et al. 2016) in helping to understand charge (de)stabilization. It is now clear that the key factors in

water chemistry that apply to natural colloids can also largely be applied to manufactured NMs. The key factors in particle aggregation of charge-stabilized NMs in environmental systems are pH, ionic strength, the presence of divalent ions, and the type/concentration of organic matter (Handy et al. 2008a; Bian et al. 2011), along with the concentration of manufactured NMs (Baalousha et al. 2016). For charge-stabilized systems, the effect of these variables is mediated by charge, so pH may lead to different behaviors, whereas increases in ionic strength will lead to aggregation. Homoaggregation/agglomeration (where agglomeration is usually defined operationally as a weaker interaction than aggregation) is commonly studied in NMs and may be essentially instantaneous after the addition of NMs to test solutions, yielding aggregate/agglomerate sizes in excess of 100 nm. Many studies have demonstrated the role of NOM such as humic and fulvic acids in stabilizing NMs against aggregation (see Domingos et al. 2009; Angel et al. 2013; Yang et al. 2017), through both charge and steric repulsion.

The importance of heteroaggregation (aggregation between non-similar particles) at high NM concentrations was demonstrated by Quik et al. (2012) using 1 mg/L of nano-CeO₂ added to filtered and unfiltered river waters. In unfiltered waters, heteroaggregation with natural colloids led to 80 to 85% removal of the ceria by sedimentation over 4 d, following first-order kinetics. By contrast, NOM in the filtered waters stabilized the NMs for up to 12 d. At higher ceria concentrations (10 and 100 mg/L in unfiltered waters), more than 99% settled out, largely by homoaggregation. In a filtered algal medium, increasing additions of NOM increasingly stabilized nano-CeO₂ by adsorption, reducing homoaggregation by increased electrosteric (i.e., combined electrostatic and steric) repulsion, as evidenced by an increasingly negative zeta potential (Quik et al. 2010); steric stabilization alone is also likely to be important (Diegoli et al. 2008). Recall that ceria concentrations are likely to be in the µg/L range for freshwaters (Figure 2), several orders of magnitude lower than the experimental concentrations in most studies. The demonstration of first-order kinetics for both sedimentation and dissolution was a useful input for future

modeling exercises; however, rate constants could not be readily estimated (Quik et al. 2011).

In a later study (Quik et al. 2014), heteroaggregation rates with natural colloids and sedimentation rates were estimated for C60, nano-CeO₂, PVP-coated nano-Ag, and SiO₂-coated nano-Ag for a range of river waters from the Netherlands. System-specific parameters such as these will be important for site-specific modeling. Such studies led to other investigations of heteroaggregation with clay particles using high concentrations (0.1 and 0.8 mg/L) of nano-TiO₂ with natural clays (Labille et al. 2015) and humic acid colloids (Praetorius et al. 2014). Although they provide useful starting data, these studies may substantially overestimate the importance of homoaggregation, given that aggregation kinetics are strongly related to the initial concentration (in particular the number concentration) of the dispersed NMs (Merrifield et al. 2017b).

The nature of the NMs in different sinks and receptors may also be fundamentally different because of aggregation and dispersion. The water column can be expected to contain smaller, dispersed NMs, whereas the benthos are exposed to aggregated NMs, at higher mass but likely lower toxicity per unit mass (Römer et al. 2013). Although aggregation often reduces bioavailability from waterborne NM exposures (Khan et al. 2012), there is no evidence that it affects bioavailability from ingested NMs (Croteau et al. 2011a, 2011b). In some cases, aggregation can enhance bioaccumulation by making particles accessible (Ward and Kach 2009) or by increasing ingestion rates (Croteau et al. 2014).

Recently the concentration dependence of aggregation (and dissolution) has been shown (Figure 3) by measurement of number concentration and NM mass (size) of core-shell NM using SP-ICP-MS (Merrifield et al. 2017c). Homoaggregation was shown to be quantitatively unimportant at realistic environmental concentrations (<1 µg/L) and relevant time-scales, suggesting that heteroaggregation may be a more important mechanism, given the higher concentrations of natural colloids.

The NMs stabilized by other mechanisms, primarily steric interactions, are largely unaffected by solution conditions such as ionic strength and are far more stable, especially in hard and marine waters (Badawy et al. 2010). Natural organic matter has been shown to provide additional colloidal stability through replacement of original coatings and subsequent electrosteric repulsion (Diegoli et al. 2008), as well as additional charge repulsion. However, sterically stabilized NMs are prone to aggregation at higher concentrations (Alabresm et al. 2017), usually significantly higher than likely environmental concentrations, possibly due to polymer entanglement and bridging mechanisms.

Both charge and steric stabilization of NMs can influence aggregation, as illustrated by El Badawy et al. (2012) for Ag NMs. Both uncoated and citrate-coated nano-Ag were stabilized by the negative surface charge on the particles caused by adsorbed hydroxyl ions and citrate molecules, showing slow aggregation over short time periods in low-ionic-strength (10 mM) NaCl or NaNO₃ solutions. Polyvinylpyrrolidone coatings sterically stabilized nano-Ag with very little change in size, even in 1 M

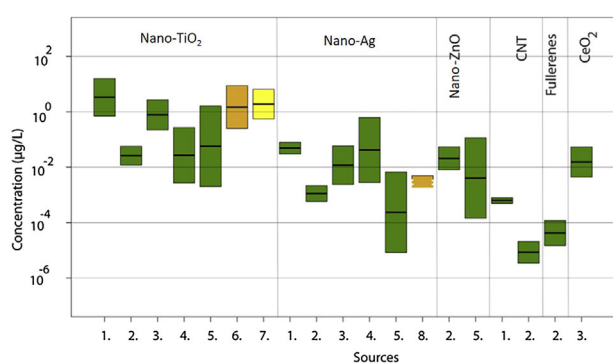


FIGURE 2: Modeled and analytical concentrations of NMs in surface waters. The green boxes show the range (and the arithmetic mean on the log scale) of modeled results. The yellow boxes show measured concentrations, and the orange boxes combine measurements and modeling. CNT = carbon nanotube. NM = nanomaterial. (From Gottschalk et al. 2013; for sources of data, see Gottschalk et al. 2013.)

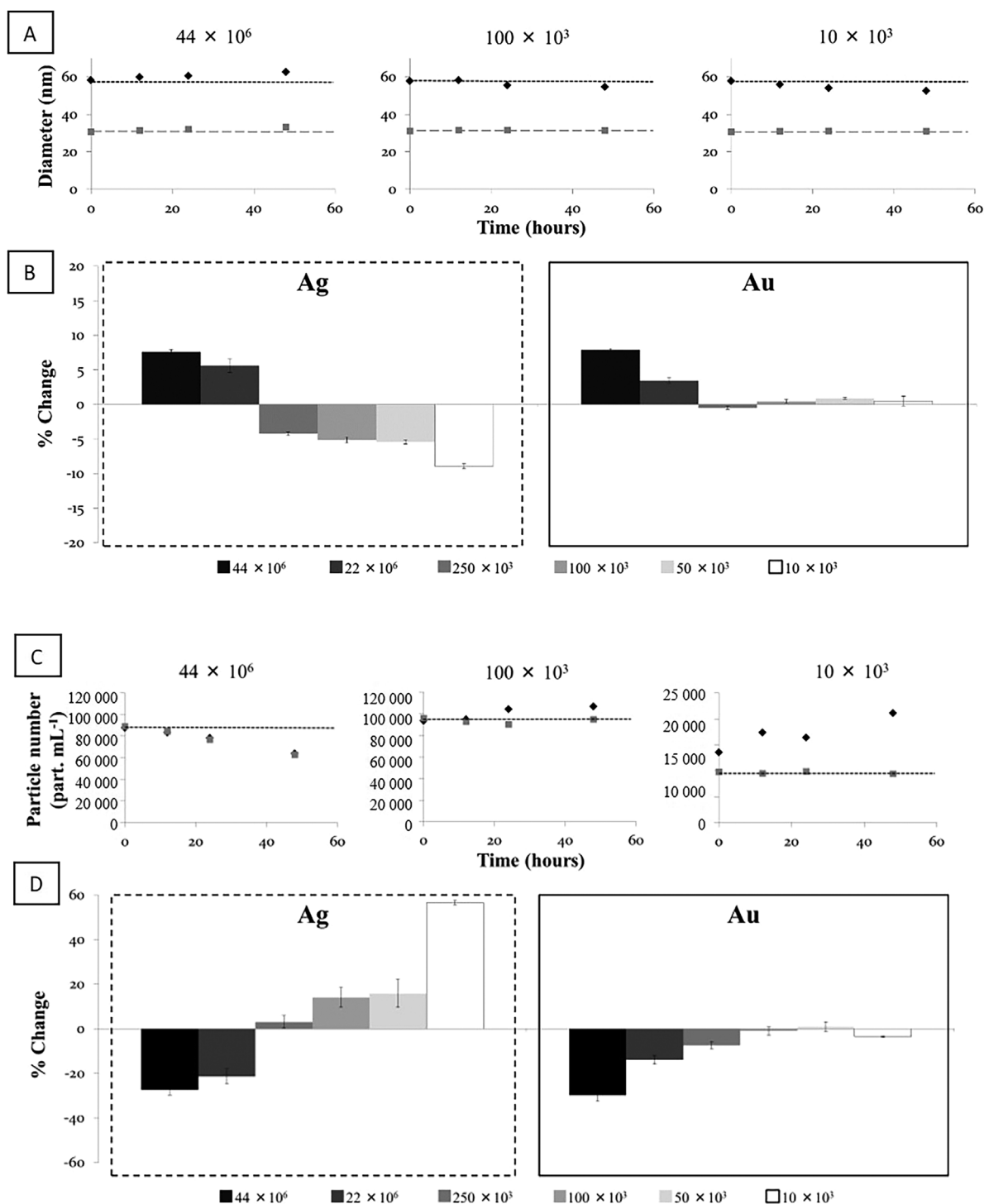


FIGURE 3: Concentration-dependent transformations to nanoparticle (NP) diameter and number concentration in a moderately hard water (MHW). (A) Particle diameter versus time for concentrations of 4.4×10^7 , 1×10^5 , and 1×10^4 particles/mL. (B) The percentage change in particle diameter of NPs in MHW after 48 h for 6 concentrations (from Merrifield et al. 2017a). (C) Particle number concentration versus time for concentrations of 4.4×10^7 , 1×10^5 , and 1×10^4 particles mL^{-1} . (D) The percentage change in particle diameter of NPs in MHW after 48 h for all 6 concentrations (4.4×10^7 , 2.2×10^7 , 2.5×10^5 , and 1×10^5 , 5×10^4 , and 1×10^4 particles/mL). (From Merrifield et al. 2017b.)

NaCl. Such effects were not predicted by the DLVO theory, of course, although there are models that can predict steric behavior. Coating with NOM also afforded stabilization due to steric repulsion (Domingos et al. 2009), although for some NMs, depending on solution pH, effects on the zeta potential were evident. Similar results have been found by others (Badawy et al.

2010), and the role of both stabilization mechanisms should be considered. The literature reveals an over-reliance on concepts of charge stabilization over steric stabilization; zeta potential is often used as a surrogate for colloidal stability, for instance. In addition, although increased colloidal stability is usually found and discussed, it does not always occur, and NOM under certain

conditions can cause aggregation (Baalousha et al. 2008; Omar et al. 2014). Understanding the conditions under which NOM increases stabilization or destabilization, and the mechanism by which this is effected, is a key point in colloidal stability in environmental systems.

Sulfidation and redox behavior

Sulfidation is a major chemical transformation for many metal NMs, particularly in the presence of enhanced sulfide concentrations such as those found in parts of wastewater treatment plants (WWTPs) or in anoxic or sub-oxic sediments (Kim et al. 2010; Kaegi et al. 2011). The reactions can result in changes in particle size, surface charge, and solubility and are often thought to be caused by core-shell ($\text{Ag}@\text{Ag}_2\text{S}$) formation, where the Ag_2S layer gradually increases. Ultimately these changes will influence the fate, bioavailability, and effects of the NMs.

The identification of Ag sulfide (Ag_2S) NMs in sewage sludge (Kim et al. 2010) provided field evidence of sulfidation of the widely used Ag NMs, and microcosm studies have shown that sulfidation was occurring (Auvinen et al. 2017). The reaction mechanism requires both oxygen and sulfide and may be either a fast, direct surface reaction or a slower, indirect reaction involving release of ionic Ag and more rapid precipitation of Ag_2S (Liu et al. 2011; Levard et al. 2013). Oxy-sulfidation is the preferred route when sulfide concentrations are high (mg/L; Liu et al. 2011), and NOM concentration plays a protective role in reducing sulfidation rates (Baalousha et al. 2015). The extreme insolubility of Ag_2S ($K_{\text{sp}} = 6 \times 10^{-51}$) means that Ag^+ will likely exchange with other less soluble sulfides (ZnS , FeS), and toxicity will generally be reduced (Devi et al. 2015), although this process may not lead to coherent core-shell NMs (Baalousha and Lead 2015). In addition, preservation of the original NMs has been observed (Baalousha et al. 2015; Pettibone and Liu 2016; Römer et al. 2016), but reversibility (e.g., via transport from sediments to overlying waters through bioperturbation) is poorly understood; examination of the literature on trace metals will be a useful starting point to increase our understanding of such processes for inorganic NMs.

Nano-ZnO can undergo slow sulfidation by a surface dissolution and reprecipitation mechanism (Ma et al. 2013). As with partially sulfidized Ag NMs, the solubility of ZnO was not quenched by a partial coating, although coating will likely play a key role in controlling dissolution rates. With increased sulfidation, solubility was reduced, although the transformation process does generate Zn^{2+} . The newly formed, sulfidized nano-ZnO particles were found to be smaller, but, with a reduced surface charge, they are often more susceptible to aggregation. Nano-CuO sulfidation also occurs over several days with the initial formation of copper (I) sulfide (Cu_2S) and possible transformation to CuS (Ma et al. 2014). Unlike nano-Ag and nano-ZnO, the sulfidized form of the original nano-CuO has greater solubility and releases more Cu^{2+} than the parent NM (Ma et al. 2014); as a consequence, the sulfidized form has been shown to be more toxic to aquatic biota (Li et al. 2015).

More generally, oxidation is not a major transformation pathway for most NMs, although it is an essential step in the

dissolution of metals such as Ag, whereas redox transformations of metal oxides such as FeO and ceria are important in determining behavior. The effectiveness of Ag NMs as bactericides likely relies in part on the surface oxidation of elemental Ag, and recent studies suggest that photochemical oxidation can enhance the formation of a surface layer of Ag^+ (Grillet et al. 2013), with NOM again having a protective effect on Ag NMs (Römer et al. 2016).

Modeling the exposure and fate of NMs in the environment

Fate and behavior considerations necessarily start with discharges to the environment, which are known in principle but very poorly known in practice. A conceptual model of the routes into and through the environment for NMs is shown in Figure 4. There are limited, hard-to-find, and poor-quality data on who is manufacturing, using, and discharging NMs and in what amounts. Regulations such as the European Union's Registration, Evaluation, Authorisation and Restriction of Chemicals and the US Environmental Protection Agency's Toxic Substances Control Act should improve this situation, at least potentially, by making these data more publicly available. The main discharge routes include point or diffuse sources to freshwaters, including treated wastewater, sludge application to soils, and landfill leachate. Other primary sources of NMs into the environment include emissions associated with production, spillage during transportation, handling, and storage, and discharges associated with waste handling.

Increasing numbers of publications have been directed toward modeling the concentrations and fate of NMs entering the environment. The current status of the different forms of modeling and the differences in their underlying assumptions have been reviewed in recent publications by Gottschalk et al. (2013) and Baalousha et al. (2016), with distinctions drawn between mass flow analysis and fate and behavior models. The former tend to concentrate on providing input data, whereas the latter tend to focus on within-environment processes such as aggregation. Although computationally challenging, the nesting of these models such that the mass flow analysis models provide input parameters to the fate and behavior models would be ideal and has already partly begun (Baalousha et al. 2016).

A major limitation of the mass flow analysis models is still a lack of adequate input data. Models clearly require data on NM production and usage in industrial and consumer products. Subsequently, there needs to be an evaluation of the extent to which NMs in these products are released to the environment (Wohlleben and Neubauer 2016); in addition, calculation of release rates and later transformation rates and types are needed. The lack of analytical capabilities for detection and quantification of NMs in real environmental systems currently also makes validation against actual environmental concentrations impossible (Nowack et al. 2015). Input data for the mass flow analysis models must be laboriously collected (Mahapatra et al. 2015) or must be estimated within quite wide bounds. The outputs from these models are therefore not definitive, as they are often portrayed, and should be used as a guide and with circumspection.

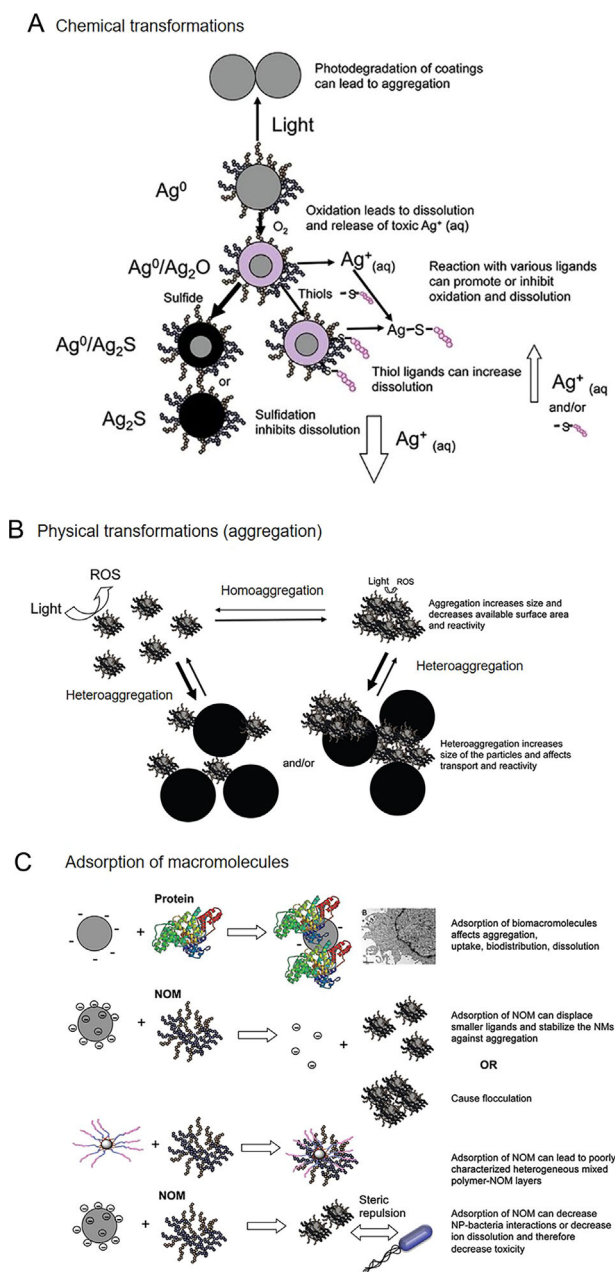


FIGURE 4: Conceptual diagram of the major transformations that engineered nanomaterials (ENMs) might undergo in the environment. (Modified from Lowry et al. 2012a.)

The earliest modeling by Boxall et al. (2007) relied on global production data, with a focus on products having free, engineered NMs in such product categories as cosmetics, paints and coatings, catalysts and lubricants, water treatment, food and food packaging, human and veterinary medicines, and plant protection products. Mueller and Nowack (2008), in considering environmental impacts in Switzerland, used global production estimates converted on the basis of the Swiss population compared with that of the industrialized world, but there were clear limitations in approaches in large part due to the difficulty of gathering reliable data. As newer data on production volumes became available and as models became more

sophisticated, more reliable estimates have been achieved (Keller et al. 2013; Sun et al. 2014). For instance, global production data, regional projections, and information on local production were used (Gottschalk et al. 2009, 2011; Keller and Lazareva 2014; Sun et al. 2014). In addition, models developed probabilistic approaches that considered the distribution of concentrations at various stages of the material flow analysis (Gottschalk et al. 2009, 2010; Sun et al. 2014) to account for the large uncertainties and variability in model input parameters.

A generalized structure of a mass flow analysis model is shown in Figure 4 (Sun et al. 2014): the releases from primary sources go principally to elements of the engineered environment, namely WWTPs, waste incineration plants, landfill, or recycling, and also directly to the natural environment (air, soil, water, and sediment). There is transfer from these 2 broad compartments as well as transport within each of these compartments. For instance, aggregation and settling will remove NMs from the aqueous phase into the sediments, with likely resuspension in many cases. The WWTPs are often generally assumed to be the major recipient of many NMs, although a number of other environmental receptors exist because of significant misconnection of drains (Mahapatra et al. 2015) and other sources.

Fate processes were largely ignored in the earlier studies (Boxall et al. 2007). Bottom-up approaches (Mueller and Nowack 2008) are more fully life-cycle based, and recent fate and behavior models include more detailed processes such as dissolution, sedimentation, and aggregation, often linked to stream flow and other physical processes (Praetorius et al. 2012; Liu and Cohen 2014; Sun et al. 2014; Dale et al. 2015; Sani-Kast et al. 2015; Ellis et al. 2016, 2018). Despite these advances, many uncertainties and deficiencies remain. Most models assume steady-state concentrations in various compartments, but variability in the dynamics of transformation is very important and not well understood.

In addition to the lack of methods for the analysis of environmental concentrations, laboratory studies of NM fate are usually undertaken under simplified conditions, using concentrations much higher than those expected in the environment. The concentration effect on dissolution, aggregation, and more complex transformations has recently been quantified for Ag NMs (Hadioui et al. 2013; Baalousha et al. 2016), whereby a dissolution-dominated regime occurs below $1 \mu\text{g/L}$ and an aggregation-dominated regime occurs above $10 \mu\text{g/L}$ for Ag NMs (Merrifield et al. 2017a). This change has consequences for bioavailability, which is also concentration dependent (Croteau et al. 2014). The effects are likely mediated through changes in aggregation behavior. Similarly, unusual behavior of the diffusion coefficients of TiO_2 NMs with concentration may be related to aggregation (Holmberg et al. 2011).

Predicted environmental concentrations

As mentioned, analytical measurements taken in real-world systems are almost nonexistent. Because of analytical challenges, detecting and quantifying NMs in real environmental systems remain essentially impossible. As a result, there are few

data on actual concentrations against which to validate fate and behavior and mass flow analysis models (Nowack et al. 2015). In assessing NM risk to the environment, probable (or predicted) environmental concentrations (PECs) are required that can be compared with predicted no-effect concentrations (PNECs). In the absence of measured data, modeling has been our sole method of providing estimates of exposure concentrations. Examples of PECs for NMs in freshwaters determined from a range of modeling approaches, together with measured data, are shown in Figure 2. Note that the examples provided are for a freshwater compartment, but similar estimates have been made for soils, sediments, and wastewater treatment effluents and sludge, as reviewed in many of the studies cited elsewhere in the present article. Even for these freshwater systems, the limited number of data sets shows a wide range of values, possibly reflecting real differences in the environment, differences in the methodological approaches, or both. Ranges of up to 4 orders of magnitude for nano-TiO₂ and nano-ZnO, and up to 2 orders for nano-Ag and CNTs are shown in Figure 2.

It has been suggested (Gottschalk and Nowak 2012) that models based on top-down approaches can easily be in error because they rely on imprecise estimates of market penetration. Nevertheless, estimates based on bottom-up modeling at a local scale can be equally variable, with ranges such as 11 to 1600 and 4 to 320 ng/L being estimated for TiO₂ NMs and Ag NMs, respectively (Gottschalk and Nowack 2012), and upper values exceeding measured concentrations (Gottschalk et al. 2013). Recent modeling of releases to European rivers of nano-ZnO and nano-Ag (Dumont et al. 2015) found that half of the river stretches had predicted long-term (months to years) average concentrations exceeding 0.002 ng/L for nano-Ag and 1.5 ng/L for nano-ZnO; the authors noted that these values were based on only household products and so are likely underestimates of environmental concentrations. In addition, these values were based on recent estimates of discharges of NMs, not on likely increased future discharges, again underestimating likely concentrations in the long term.

This large variability has disturbing implications for the reliability of risk estimates. Importantly, it reinforces the need, already noted by several authors (Gottschalk and Nowack 2012; Gottschalk et al. 2013), for more localized monitoring of NMs in waters, sediments, and soils, provided that appropriately sensitive analytical methods are available.

Fate and behavior of NMs in marine ecosystems

The earlier discussion in this section on Processes and transformations affecting NM fate and behavior in surface waters, specifically refers to freshwater systems; the operative processes are similar for marine systems, and we now discuss the differences between the 2 systems. Theoretical concerns center on the higher ionic strength (and to some extent on lower NOM concentrations) of marine systems, which would lead to charge screening, aggregation, and particle settling, for charge-stabilized NMs (Klaine et al. 2008). Thus, coastal sediments and those in the deep ocean are considered likely sinks (Klaine et al. 2008). However, microbial and physicochemical activity in

marine sediments will likely transform NMs and/or remobilize them into the water column. There is little direct field evidence, but mesocosm and laboratory studies and read-across from other contaminant behaviors suggest this is likely (Galloway et al. 2010). In addition, near-shore environments might be expected to have higher concentrations than the open ocean given the likely terrestrial discharge sites, although again there are few or no direct data. The viscous properties of ultrafine particles in the sea-surface microlayer are also a concern and may be an important sink for certain types of NMs. Little progress has been made in this area either.

Progress has been made on understanding the settling rates of NMs in saline conditions. Some of this knowledge is derived from studies on physiological salines that also have relatively high salinities. For example, Al-Jubory and Handy (2013) showed that artificial gut saline for trout rapidly precipitated TiO₂ particles in a size-dependent manner, leaving only the smallest size fraction (mainly primary particles) in suspension; sterically stabilized NMs are often far less susceptible to aggregation and are more likely to remain in the aqueous phase (Hitchman et al. 2013). Particle settling rates have also been measured in seawater (nano-TiO₂: Brunelli et al. 2013; C60, nano-CeO₂, and various nano-Ag-containing particles: Quik et al. 2014), almost always at high concentrations. The settling behavior is dependent on the particle type, with aggregation kinetics increasing at higher mass concentrations of each material, in general agreement with the DLVO theory. However, for nano-TiO₂, at least in one study, the settling rates in seawater appeared to be similar or only a little faster than those in artificial freshwater (Brunelli et al. 2013).

The accumulation of chemicals in marine and estuarine (and freshwater) sediments is generally well known (Honeyman and Santschi 1991). However, details of the sedimentation behavior of NMs are poorly quantified, particularly at environmentally relevant concentrations, and in mixing zones in freshwater, seawater, and estuaries. In addition, how NMs would be altered by these interactions and the fate of the NMs within these processes are largely unknown. However, it is hypothesized that NMs in the porewater of sediments will behave similarly to NMs in the overlying water, and the binding of NMs to natural minerals and organic matter in the structure of the sediment will be broadly similar to that of natural colloids and other chemicals. For example, dissolution and complexation processes should still occur in the sediment porewater, and the mobility of NMs in the sediment will depend on the sediment grain size and its chemical composition, temperature, and salinity, as well as the effects of bioturbation.

There are only a few studies on the behavior of NMs in marine sediments. Bradford et al. (2009) showed that serially dosing intact sediment samples from Plymouth Sound in the United Kingdom with Ag NMs to simulate a daily effluent discharge with the tide, resulted in rapid loss of Ag metal from the water column and accumulation in the top 1 cm of the sediment. The total Ag remained trapped in the surface without transfer to deeper parts of the sediment. There were also no discernible effects on the microbial biodiversity in the sediments based on molecular

methods such as density-gradient gel electrophoresis of the nucleotides. A follow-up study on Plymouth Sound sediment showed that the antibiotic resistance of the microbes was also unaffected (Mühling et al. 2009). At high concentrations, nitrification processes have been reduced over short-term studies, but not at lower concentrations over similar timescales (Beddow et al. 2017). In freshwater sediments, the cumulative respiration rate and microbial biomass of the sediment was unaffected by exposure via the overlying water containing up to 250 mg/L of Ag as Ag NMs (Colman et al. 2012). In the same study, the freshwater sediments showed normal microbial services with no loss of sulfatase, phosphatase, or luciferase activity. Such detailed studies of ecosystem services from microbes are still lacking for Ag NMs in marine sediments. In addition, the limited data sets available show somewhat contradictory conclusions that need to be rationalized by appropriate choice of NMs, conditions, and endpoints in further studies.

Mesocosm data have shown that Au NMs are present mainly in sediment biofilms in estuarine systems (Ferry et al. 2009). Marine sediments are also subject to bioturbation by polychaete worms and shellfish living on or in the sea floor. A quantitative analysis of how bioturbation alters the fate and behavior of NMs in marine systems has yet to be performed, but it is clear that the behavior of marine organisms in sediment can sometimes be affected after NM exposure. For example, polychaete worms can show a decrease in casting rates (nano-TiO₂: Galloway et al. 2010), or, alternatively, burrowing behaviors may not change very much (nano-CuO: Buffet et al. 2013). Similar observations are also being made for sediments in freshwater lakes. Pakarinen et al. (2014) showed that up to 9% of fullerenes deposited in surface lake sediments were remobilized over several days as a consequence of surface currents, as well as by bioturbation in the natural environment. However, the rates of bioturbation along with tidal flows and wave action suggest that such mixing and remobilization effects will be much higher in coastal marine ecosystems.

Fate and behavior in terrestrial systems

A number of processes that govern the fate of NMs in soils are similar to those that govern fate in aquatic systems, namely, dissolution, transformation, and aggregation/disaggregation. However, some of the processes governing NM fate and behavior in soils are quite different, for example, straining, deposition/mobilization, and diffusive transport (Figure 5). These vary in importance depending on the characteristics of both the NM and the soils (Cornelis et al. 2011, 2014).

For some NMs, dissolution may be very important, because it degrades them so that fate and bioavailability become more aligned with the soluble components. For example, it has been demonstrated that bulk ZnO dissolves rapidly in soils (Smolders and Degryse 2002; McBeath and McLaughlin 2014), so that ZnO NMs are also likely to be ephemeral in soils unless coated with agents to restrict dissolution. Heggelund et al. (2014) showed an absence of NP-specific effects in soils, with bulk and nano-ZnO behaving similarly in terms of fate, toxicity, and dissolution, with

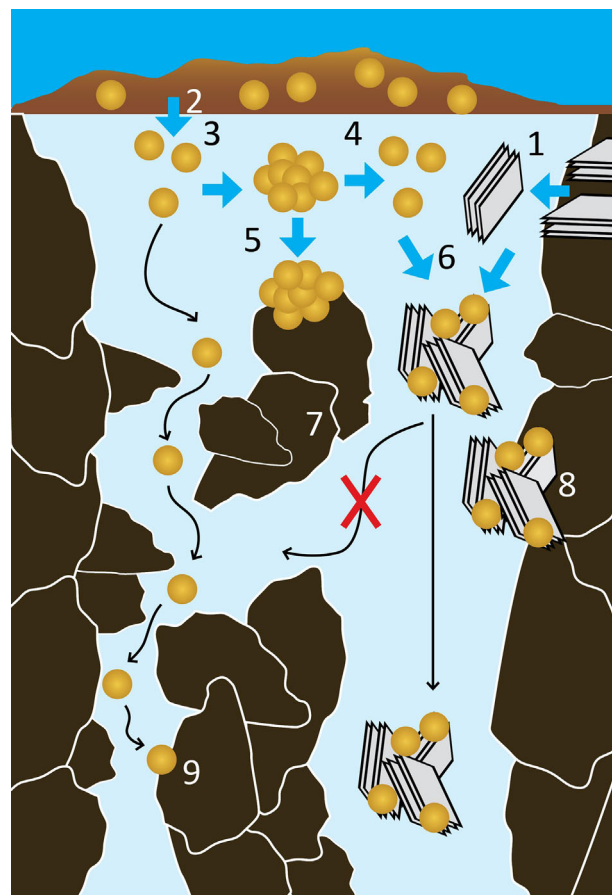


FIGURE 5: The main fate-determining processes for natural colloids and nanomaterials in soils. 1, colloid generation; 2, engineered nanomaterial leaching from biosolids; 3, homoaggregation; 4, fragmentation; 5, sedimentation; 6, heteroaggregation; 7, size exclusion; 8, straining; 9, deposition; 10, convective transport. (Adapted from Cornelis et al. 2014.)

pH being the most important control on dissolution. The ZnO NMs can dissolve and/or be transformed into a mixture of species such as ZnS, Zn₃(PO₄)₂, Zn-cysteine, Zn-substituted ferrihydrite, and Zn²⁺ adsorbed to mineral surfaces. The reported proportions of these components will be different depending on the wastewater treatment configuration (Lombi et al. 2012; Ma et al. 2013; Wang P et al. 2013; Brunetti et al. 2015; Judy et al. 2015b).

It is worth noting that dissolution of NMs in soils is not easily predicted by batch water solubility tests, because soil surfaces provide a sink for anions or cations released from the NMs (promoting dissolution), and also provide strong pH buffering of the soil solution. Thus dissolution of metal oxides is much faster in soils than in water. Carbon-based NMs are generally insoluble (in water) but may degrade through photolysis or microbial decomposition (Chouhan et al. 2016; Navarro et al. 2016).

In a recent review of the fate of NMs in soils, Cornelis et al. (2014) performed a useful comparison of typical pH and ionic strengths of soil saturation extracts (using a global database) in relation to typical critical coagulation concentrations for NMs. They concluded that homoaggregation of NMs would be slow in most soils because the pH and ionic strength of most soil

solutions would lie below the critical coagulation concentration of most NMs. Heteroaggregation is likely to be very important in soils, as in aquatic environments, because soil porewaters often contain higher concentrations of natural colloids in suspension. Numerous studies have observed strong heteroaggregation of NMs with soil colloids (Cornelis et al. 2010, 2011, 2012; Hotze et al. 2010; Huynh et al. 2012; Hoppe et al. 2015; Klitzke et al. 2015; Labille et al. 2015; Smith et al. 2015), which has significant implications for limiting NM transport through soils because straining will be enhanced (Figure 5). On the other hand, the presence of NOM in soil porewaters has often been found to stabilize NMs and inhibit both homo- and heteroaggregation (Praetorius et al. 2014). This means that for the less soluble NMs, accumulation will likely occur in topsoils with little movement to depth in most soils. Few studies have examined transport under field conditions (Kasel et al. 2013), and this is probably the key gap for more accurate assessment of the real risk of NM transport through soils.

Over the last decade, studies of NM transport through soils have progressed from using inert stationary phases (e.g., quartz beads) in columns (Lecoanet et al. 2004) to the use of natural soils, so that we now have a much better appreciation of the potential transport of NMs in terrestrial systems (Jaisi et al. 2008; Darlington et al. 2009; Fang et al. 2009; Jaisi and Elimelech 2009; Wang et al. 2010; Wang DJ et al. 2015; Cornelis et al. 2012, 2013; Coutris et al. 2012; Sagee et al. 2012). The CNTs appear to be retained in soils due to their high aspect ratio, leading to significant straining (Jaisi and Elimelech 2009; Kasel et al. 2013). Fullerenes are also strongly retained in soils, likely through strong interactions with soil organic matter (Wang et al. 2010; Navarro et al. 2013). Where some mobility was observed, this was usually in pure sand media or very sandy soils with very low organic matter content (Zhang LL et al. 2012).

NM BIOAVAILABILITY AND BIOACCUMULATION IN AQUATIC ORGANISMS

Studies on bioavailability and uptake are critically important to link the environmental chemistry of NMs to biological effects. The assumption is that the presence of a NM on or in an organism will lead to a biological response, and this can be informed by how the NM in the environmental media initially interacts with the external surfaces of the organism. In 2008, it was quickly identified that the broad concept of substance behavior in water, adsorption of a bioavailable fraction to the epithelial surface of the organism, and then membrane uptake to internalize the substance could be applied to NMs (Handy et al. 2008a, 2008b; Klaine et al. 2008). The steps in the net uptake or absorption to the internal body fluids are summarized in Figure 6. The key steps involve how the NM behaves in the external media (e.g., water or gut lumen) and is presented to the external surface of the organism. The latter is a dynamic microenvironment where secretions from the organism can interact with the external media and/or act as a concentrating layer for the substance. Transformation processes similar to those that occur in water and soil might also occur. Figure 6

shows the uptake across an idealized epithelium such as a fish gill, but conceptually many organisms have uptake pathways for solutes (metal transporters are highly conserved across species) and also for particulates via endocytosis pathways. The experimental evidence for the bioavailability and the uptake mechanisms of different NMs in aquatic organisms is far from complete, and there are inherent differences in how, for example, invertebrates process metal particles compared with fishes.

Bioavailability and uptake studies in invertebrates

Effect studies largely dominate the scientific literature on aquatic invertebrates and engineered NMs (Selck et al. 2016). Although important, these studies provide a limited understanding of the processes linking the sources and transformations to bioaccumulation and ultimately toxicity. Fewer studies have addressed NM bioaccumulation, for which bioavailability is a driver. The ability of aquatic invertebrates to accumulate NMs has been unequivocally demonstrated (Garcá-Alonso et al. 2011; Khan et al. 2015). Clearly, NMs provide a unique type of exposure that is not fully considered in the risk assessment process for metals alone (Luoma et al. 2014). For instance, the internalization of NMs is poorly understood, although much progress has been made since 2008, and there is significant evidence of nano-specific effects.

The properties and behaviors of NMs are important drivers of bioaccumulation in invertebrates. For example, particle size has been shown to influence bioaccumulation, although NM size may not be indicative of exposure if particles aggregate. Many studies have shown that bulk or micron-size particles are less bioavailable to invertebrates than their dissolved or nanosized counterparts (Pang et al. 2013; Cozzari et al. 2015). Numerous studies have shown that metal uptake rates are faster for ionic forms than for nanosized forms (Zhao and Wang 2010; Croteau et al. 2011a, 2014; Ramskov et al. 2015). For example, Ag uptake rates were 2 to 10 times faster for dissolved Ag than for Ag in nanoparticulate forms for the estuarine snail *Peringia ulvae* (Khan et al. 2012), the freshwater snail *Lymnaea stagnalis* (Croteau et al. 2011b), the water flea *Daphnia magna*, and the freshwater oligochaete *Lumbriculus variegatus* (Khan et al. 2015).

Particle composition also has an important influence on NM bioaccumulation in invertebrates. In general, Au, TiO₂, and SiO₂ NMs were less bioavailable and toxic than CuO, ZnO, or Ag NMs (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript). For example, *D. magna* can efficiently ingest Au NMs, but its gastrointestinal tract can be largely purged after 1 h of depuration in clean water in the presence of food (Khan et al. 2014). In contrast, citrate-capped Ag NMs accumulated after ingestion of diatoms mixed with NMs were retained in the tissues of the snail *L. stagnalis* with no detectable loss after transfer to clean media for up to 14 d (Croteau et al. 2011b). Likewise, the Cd accumulated after ingestion of quantum dots (CdS and CdSe) was retained in the tissues with negligible elimination (Khan et al. 2013b). Comparison of data from studies conducted with different approaches and particle

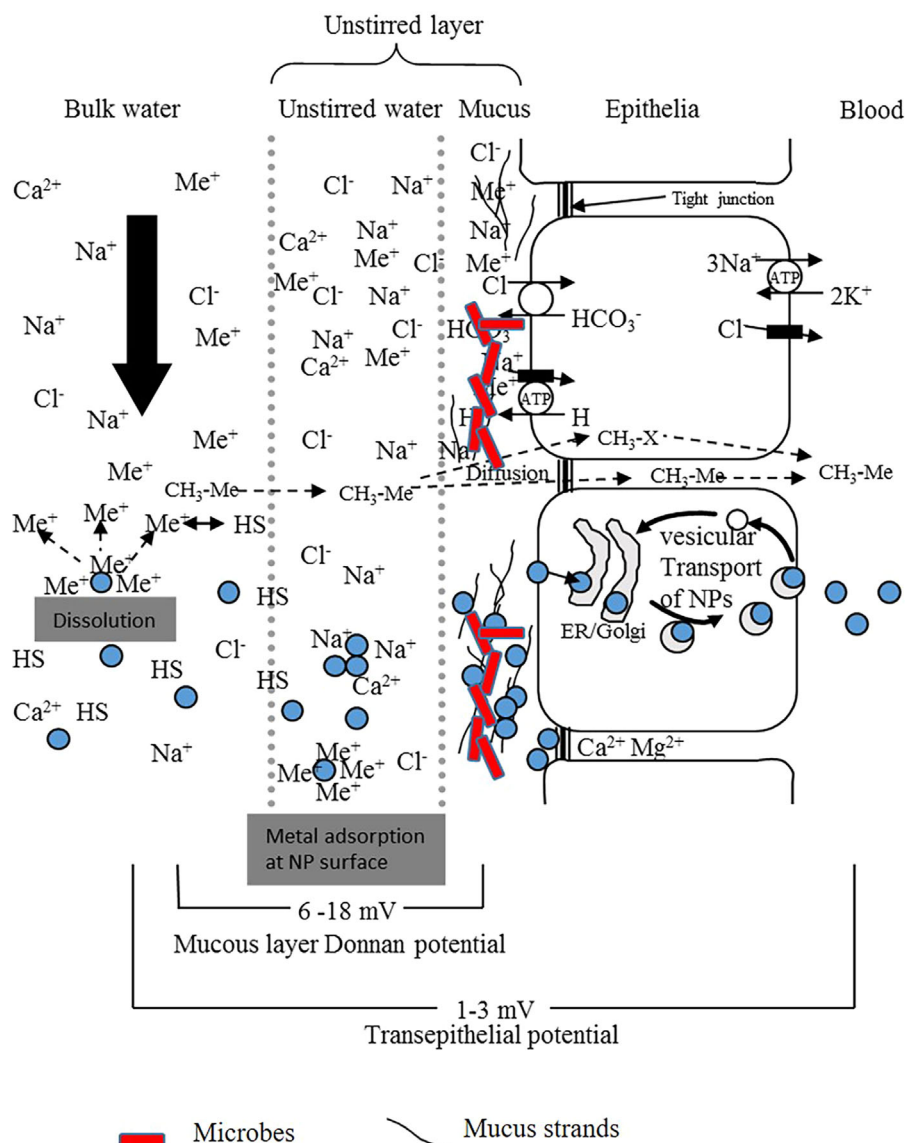


FIGURE 6: An idealized diagram of an epithelium (freshwater fish gill) showing the mechanisms of uptake for electrolytes, toxic metal ions (Me^+), and electroneutral diffusion of some small organo-metals ($\text{CH}_3\text{-Me}$), compared with nanoparticles (NPs; filled circles). The substances in the bulk solution (i.e., the external environment) must diffuse into an unstirred layer (USL) comprised of water/mucus secretions and microbial biofilm, prior to transfer into the epithelium itself. The upper portion of the diagram shows electrolytes and toxic metal ions that diffuse into the USL, and which may bind to strands of mucus (mostly polyanionic) where the exclusion of free anions like Cl^- from the mucus layer contributes to the Donnan potential at the apical surface. Electrolytes and toxic metal ions usually move through the cell using ion transport pathways (Na^+ transporters are illustrated here). The NPs will diffuse into the USL, albeit at a slower rate than smaller molecules or solutes, and may be influenced by humic substances (HS). Cationic NPs will bind to strands of mucus by electrostatic attraction, but (regardless of surface charge) they may also become entangled in the mucoproteins (steric hindrance) to prevent uptake by the epithelial cells. The NPs are too large to be taken up by ion or other transporters on the cell membranes, although diffusion is known for small lipophilic NPs. The Ca^{2+} - and Mg^{2+} -rich environment in the tight junctions suggests that NPs would aggregate rather than diffuse through the paracellular route. In addition, some nanometals may release free metal ion (Me^+) by dissolution of ions into the bulk solution. In contrast, nanomaterials can also show surface adsorption of metals, and this is likely to be faster in the higher ionic strength of the USL. Diffusion of charged NPs into the USL will be affected by the Donnan and transepithelial potentials, in a similar way to other charged substances. Uptake of NPs through vesicular transport has been pharmacologically confirmed for some engineered nanomaterials. ER = endoplasmic reticulum; ATP = adenosine triphosphate. (Modified from Handy et al. 2008b.)

properties should be performed with due care; however, these results show that a lack of (or slow) elimination of metal-based NMs has important implications for bioaccumulation: slow metal efflux rates of NMs will typically cause high accumulation of metals within cells or organisms.

In addition to particle size and composition, the shape of the NMs (e.g., the rods, spheres, and platelets of CuO NMs; Dai et al.

2015; Ramskov et al. 2015), the synthesis method, and the nature of the polymer used to stabilize the NM can affect bioaccumulation. These influences are much less studied partly because of the difficulty in constraining one NM feature (size, shape, etc.) while maintaining constant all other NM properties that might affect uptake and toxicity. Collaboration between materials scientists and toxicologists is still needed to fill this knowledge gap.

Bioaccumulation of NMs is further influenced by the behavior of the NM in the environment. Dissolution, for instance, may contribute to the total uptake of metal from NM exposure. This was shown, for example, in aquatic snails exposed to nano-Ag (Croteau et al. 2014; Khan et al. 2015; Stoiber et al. 2015), nano-CuO (Croteau et al. 2014), and nano-ZnO (Khan et al. 2013a). However, in most cases, bioavailability (and/or toxicity) cannot be explained solely by the metal concentrations in the dissolved phase. Aggregation can also influence bioavailability. For example, aggregation can change the dominant exposure route from water to sediment (or food) by removing NMs from the water column. As a result, potential impacts are shifted from pelagic to benthic organisms (Selck et al. 2016). However, aggregation does not eliminate bioavailability. Aggregated NMs appear to be bioavailable when accidentally ingested by deposit-feeders and grazers. For example, when assimilation efficiency was used as a measure of bioavailability from diet, the bioavailability of aggregates (or agglomerates) of NMs associated with the food ingested by the snail *L. stagnalis* ranged from 49 to 58% for Ag NMs (Croteau et al. 2011b), from 41 to 83% for CuO NMs (Croteau et al. 2014), and was 80% for ZnO NMs (Croteau et al. 2011a). Water hardness did not influence the dietary bioavailability of NMs (López-Serrano et al. 2014). In some cases, aggregation of NMs can even enhance bioavailability by forming larger particles that are preferentially retained by filter-feeding invertebrates such as mussels (Ward and Kach 2009). To the extent that bioaccumulation and toxicity are linked, exposure to highly bioaccumulative NMs is likely to elicit adverse effects more readily than exposure to other NMs.

Bioavailability and uptake studies with fishes

Similar to the studies on invertebrates, aspects of the water chemistry are known or expected to influence NM bioavailability to the gills of fishes. There are several explanations as to why a NM in the water column may become bioavailable to a fish gill (for a review, see Handy et al. 2008b). These include particle mobility—a stable dispersion of primary particles may collide frequently with the epithelium, resulting in attachment of particles to the membrane surface. The attachment and transport mechanism might be related to steric factors (shape of the NM), charge or diffusive hindrance in the mucus layer of fish and other organisms (Smith et al. 2007), or electrostatic attraction of positively charged particles to the polyanionic epithelium (although in practice most NMs are negatively charged and there will be charge repulsion). Alternatively, an unstable dispersion forming larger aggregates may simply precipitate onto the epithelium (e.g., TiO₂ particles; Johnston et al. 2010). Finally, particles may also dissolve (e.g., Cu NPs in acidic freshwater; Al-Bairuty et al. 2016) and be taken up according to traditional free ion activity models and the affinity of the dissolved form for solute transporters (Figure 6). Depending on coating and solution conditions, Cu NPs can form reasonably dispersed phases in freshwater (Shaw et al. 2016) with greater uptake from the more highly dispersed phases. However, particle settling and the greater mass concentration in larger particles or aggregates best explain

TiO₂ accumulation in/on the gut epithelium of trout (Al-Jubory and Handy 2013), whereas titania can also be found in the water column embedded with mucus produced by fish (Johnston et al. 2010).

The evidence for uptake in the particulate form versus the ionic or dissolved form of a NM is often circumstantial in fish, unlike in invertebrates, based on the dissolution behavior of the material in water or the differences in total metal accumulation in the gill between animals exposed to metal salts or the equivalent NMs. For example, with CuO NPs, the dissolution is a small fraction of the total metal in the particle dispersion in freshwater (Al-Bairuty et al. 2016), and thus it might be argued that the metal is initially taken up mainly in the particulate form.

The uptake mechanism can also be investigated pharmacologically in these larger vertebrate animals. In trout intestines, the apparent epithelial uptake is blocked by both ion transport inhibitors and agents that interfere with endocytosis pathways, demonstrating that both solute and particle transports are involved in the absorption mechanism (Al-Jubory and Handy 2013). Proof can be found, at least qualitatively, in the identification of intact particles inside the epithelial cells, although the formation of NMs in vivo cannot be discounted. For example, many aquatic species, especially shellfish, use the biomineralization processes as part of their normal biology for the formation of shell and the sequestering of metals in the tissues (Brown 1982). Metal granules can also be observed in the livers of fishes (Lanno et al. 1987). There is therefore a need to differentiate manufactured NMs from the particles already present in the tissue. This is difficult, but the availability of isotopically labeled NMs and MS-based approaches are currently yielding important results (Thit et al. 2015).

In addition, techniques available to identify and partially quantify such processes include electron microscopy coupled with energy-dispersive X-ray spectroscopy, electron energy loss spectroscopy, or other measurements for determining percentage elemental composition (for a review, see Handy et al. 2012b) as well as speciation (Merrifield et al. 2017c). Such methods allow direct determination of the particles in the tissue. However, there is a need to consider the prospect of finding small numbers of NMs in a grid made from only a handful of cells from the original tissue sample. For example, the gill surface area of a 10-g teleost fish is approximately 10 000 mm² (Hughes and Al-Kadhomy 1988), and, with a profile area of a typical cell in the epithelium of approximately 300 μm² (Laurent and Hebibi 1989), one might estimate some 3.3×10^6 cells facing the external media in a single fish. For TiO₂ NMs in freshwater, a typical 1-mg/L dispersion contains approximately 10⁶ particles/mL (Ramsden et al. 2013) of which, at best, less than one-third would be bioavailable (due to settling, losses to mucus secretion, etc.). This equates to 0.2×10^6 particles in a freshly dosed 20-L fish tank, and with typically 15 fish, only 5×10^7 gill cells would be available, or one particle in every 250 cells. Thus, even at relatively high concentrations, the probability of visualizing even a single NM in an electron micrograph of a gill epithelial cell is remote. Consequently, microscopy is probably not a useful tool for determining the presence of particles in the tissue because the risk of false negatives is high.

To enable particle detection in tissues after digestion of the tissue in strong alkali (Gray et al. 2013), SP-ICP-MS is being developed. More recently, direct measurements have been performed by single-cell (SC)-ICP-MS (Merrifield et al. 2018) at sub- $\mu\text{g/L}$ aqueous concentrations with 0 to 3 NMs/cell (in algae), in agreement with indirect measurements by Piccapietra et al. (2012). Application to fish has not been performed but is feasible. In addition, targeted Raman spectroscopy has been able to identify intact particles in or on the surface of fish gills from waterborne exposures (Johnston et al. 2010). Furthermore, studies with cultured mammalian epithelial cells show, in principle, that intact NMs can be taken up by endocytosis mechanisms (e.g., Caco-2 cells: Busch et al. 2011; Gitrowski et al. 2014). Evidence is also mounting for fish epithelial cells from gills (Felix et al. 2017) and intestines (Geppert et al. 2016), where particle internalization was reduced at low temperature, indicating an energy-dependent uptake process. However, pharmacological proof of the uptake mechanisms in the gills of aquatic organisms such as fish and bivalves is largely absent.

Dietary exposures

Hou et al. (2013) reviewed the existing literature on NP bioaccumulation by invertebrates and fish. Of 65 reviewed studies, half dealt with aquatic invertebrates. The pelagic crustacean *Daphnia* sp. was by far the most studied taxon. However, sediments are a likely sink for NMs because of the settling behaviors of particles. Benthic organisms such as worms, insect larvae, and mollusks are thus expected to be more exposed to NMs than are pelagic species (Selck et al. 2016). However, sediment exposure is understudied compared to waterborne exposure studies conducted with pelagic taxa such as water fleas (Hou et al. 2013). The difficulty in separating the NM contribution to bioaccumulation from that of background largely explains the paucity of dietary studies. For example, distinguishing newly accumulated Cu or Zn from background levels in organisms is difficult unless extremely high (and thus environmentally irrelevant) exposure concentrations are used (Dybowska et al. 2011; Misra et al. 2012). The application of tracer techniques can overcome this complication (Zhao and Wang 2010; Croteau et al. 2011a, 2014; Ramskov et al. 2015; Thit et al. 2015). For example, Croteau et al. (2014) used isotopically labeled Ag NMs to characterize Ag uptake rates in the freshwater snail *L. stagnalis* across a wide range of aqueous and dietary exposure concentrations. The results showed a concentration-dependent shift in the relative importance of dissolved Ag versus nano-Ag uptake to the snails that would not have been detectable working with unlabeled Ag. Likewise, Khan et al. (2013a) exposed estuarine snails to isotopically labeled Zn in the forms of ZnO NMs, ZnO bulk particles, and aqueous Zn, and concluded that bioaccumulation of Zn from ZnO NMs was primarily dependent on solubility. Weak Cu accumulation by the freshwater oligochaete *L. variegatus* exposed to isotopically labeled Cu (ionic Cu and CuO NMs) in water and sediment was also reported by Ramskov et al. (2015), who concluded that this species was an inefficient bioaccumulator of Cu, explaining in part the tolerance of the species. The

use of tracers to detect metals that originate from metal-based NMs is a growing field of study.

Techniques have also been developed to examine trophic transfer of NMs in the absence of solubility in the water column. Using methods developed by Cresswell et al. (2017), the uptake, assimilation, and trophic transfer of dietary nano CeO₂ particles along a freshwater food chain represented by an alga (*Pseudokirchneriella subcapitata*), a grazing snail (*Potamopyrgus antipodarum*), and a prawn (*Macrobrachium australiense*) were studied using particles containing the gamma-emitting radioisotope Ce-141 (L.A. Golding, CSIRO Land and Water, Kirrawee, NSW, Australia, unpublished manuscript). Using gamma spectrometry, pulse-chase experiments showed a rapid elimination of CeO₂ in both nano- and micron-sized forms from the digestive systems of both the snail and the prawn. More sensitive autoradiography confirmed that no Ce-141 could be detected in the prawn tissue outside the confines of the digestive tract, evidence that the particles were not transported across the gut wall and were therefore not assimilated by the biota. Nonetheless, from the perspective of predator-prey interactions, a prey item with a gut lumen full of NMs will represent a neat package of NM-contaminated food for the predator.

Dietary studies have been conducted on invertebrates and fish, although data remain sparse (Croteau et al. 2011a, 2011b, 2014; Khan et al. 2013b). Feeding studies do show that fish will eat food contaminated with NMs and continue to grow (e.g., nano-TiO₂: Ramsden et al. 2009; quantum dots: Blickley et al. 2014), but the details of changes in individual nutrients or the long-term health of the gut (or animal) are not known. Processes critical to feeding and digestive physiology are disrupted by ingestion of NMs by snails, sometimes at very low concentrations (e.g., Ag concentrations of 1–5 $\mu\text{g/g}$ in the diet: Croteau et al. 2011b, 2014). This is important because disruption of gastrointestinal function and a reduction in feeding might cause harmful changes that can affect processes such as growth and reproduction. Ultimately these changes can cause population and community changes. Uptake of metal across the gut epithelium from oral exposure to NMs has been confirmed (e.g., trout: Al-Jubory and Handy 2013). However, some unexpected observations have been made; for example, in the case of nano-TiO₂, the uptake rate across the gut of rainbow trout is dominated by a partial pressure of carbon dioxide effect (Al-Jubory and Handy 2013). The effects of barometric pressure, gas pressures, or water depth are simply not considered in environmental risk assessment for any chemicals, but this observation raises a concern that NMs may be more bioavailable at >10-m depth than at the surface. Another interesting observation has been with regard to the gut microbiome of fishes. Silver NMs especially are noted for their antimicrobial properties, but how this alters the natural microbiome of animals is not clear. Merrifield DL et al. (2013) demonstrated that the microbiome in the zebrafish gut does vary between NMs (Ag NMs and CuO NMs), but also showed a nanoeffect in that dietary CuSO₄ resulted in a microbial biodiversity different from that of CuO NMs in the fish gut. However, the toxicological or nutritional consequences of such changing microbiology of the fish gut remain to be investigated. Chronic studies at

environmentally realistic concentrations also remain a significant data gap for most aquatic organisms. A combination of isotopically labeled NMs and new methodologies models, using mesocosms, could start to address these areas in an environmentally realistic manner.

Target organs for NMs

The internal target organs for NMs in aquatic species remain poorly understood for most organisms. This is especially true of small invertebrates in which dissection for quantifying organs is impractical. In addition, the form of the materials in the internal organs (i.e., dissolved or particulate) has also not been established in the studies so far due to the technical challenges of measuring particles inside tissues. However, the microscopic imaging of invertebrates has proved useful. For example, Zhu et al. (2009) showed images of multiwalled CNTs in the gut tract of *D. magna* after a 24-h exposure. Using TEM, Heinlaan et al. (2011) demonstrated the presence of CuO NMs in the midgut of *D. magna*. Gomes et al. (2011) proposed a so-called Trojan horse delivery and release of ionic Cu following cellular internalization of CuO NMs by the marine mussel *Mytilus galloprovincialis*, similar to the suggestions for the uptake of nano-Ag and nano-CuO by human cell lines (Cronholm et al. 2013). Although microscopy approaches can demonstrate this phenomenon, the reliable identification of internalized NMs in epithelial cells from invertebrates has proved difficult. Hull et al. (2011) successfully used micro-X-ray fluorescence spectrometry to show Au NMs in the gut epithelium of the freshwater Asian clam *Corbicula fluminea*. García-Alonso et al. (2011) observed electron-dense particles resembling Ag NMs in the gut epithelia of estuarine polychaetes exposed to sediment spiked with citrate-capped Ag NMs, and the presence of Ag in these particles was confirmed by energy dispersive X-ray analysis. The SC-ICP-MS technique may prove useful here in the future.

Most of the research on internal target organs has been with fishes that are large enough to dissect. The target organs and effects on individual body systems of fishes have been reviewed (Handy et al. 2011). In general, the target organs for NMs are similar to those for their nearest equivalent traditional chemical, with the possible exception of the spleen, which appears to have an important role in processing particulate materials to prevent toxicity to the internal organs (TiO₂ NMs: Ramsden et al. 2009; Boyle et al. 2014; CuO NMs: Al-Bairuty et al. 2013). Concerns that NMs may be acutely immunogenic or neurotoxic in fishes have been proved largely unfounded from a functional perspective, although changes in the immune cells and/or white pulp of the spleen may be observed with NMs (see Al-Bairuty et al. 2013). These latter effects are mostly well within the physiological scope of the animal. So far, the potential hazards to the immune systems of fishes have been theoretically described (Handy et al. 2011; Jovanović and Palić 2012), but (for example) data on in vivo microbial challenges to the immune system during or following NM exposure are lacking. Currently there is no strong or consistent evidence of acute hypersensitivity reactions or immunosuppression in fishes exposed to NMs. Alterations in brain biochemistry (Ramsden et al. 2009) and brain

pathology (Al-Bairuty et al. 2013) have been observed in trout, but it is hard to link them to changes in animal behavior (Boyle et al. 2014). However, there have been some surprises. Nanomaterials may block the olfactory canals of fish such that they cannot chemically taste alarm substances in the water (Sovová et al. 2014). Manufactured NMs also interfere with the sensory lateral line functions of fish (McNeil et al. 2014). Such effects on their own are of limited direct toxicological consequence for the individual animal, but the ecological significance is far reaching. For example, loss of sensory perception might alter foraging behaviors, susceptibility to predation, the avoidance of contaminated water, or even fish migrations.

NM HAZARD AND ASSESSMENT IN AQUATIC ECOSYSTEMS

In the last decade, there have been numerous reviews on the ecotoxicity of NMs (Handy et al. 2008a, 2012a, 2012b; Klaine et al. 2008; Kahru and Dubourguier 2010; Fabrega et al. 2011; Levard et al. 2012; Schultz et al. 2014; Adam et al. 2015; Bour et al. 2016a, 2016b; Garner et al. 2015; Semenzin et al. 2015; Hjorth et al. 2017a). A number of systematic reviews have discussed ecotoxicity databases and the quality and regulatory adequacy of the data therein (Hartmann et al. 2017; Hjorth et al. 2017b). Unfortunately, most of the data obtained over the past decade have been for acute effects, with algal bioassays providing the majority of chronic endpoints. The acute assays have been reported largely for freshwater organisms used in Organisation for Economic Co-operation and Development (OECD) tests, often with limited dosimetry. Notably, even the OECD sponsorship program to systematically test a variety of NMs (Organisation for Economic Co-operation and Development 2007) has been criticized for a lack of endpoints, variable quality assurance on individual test methods, and substantial data gaps in toxicity such that firm conclusions could not be obtained (Hansen et al. 2017). The intention of this section is not to reiterate the findings of the acute toxicity data for NMs that have been summarized in the reviews just listed, but instead to evaluate progress on some of the data gaps that are especially important for hazard assessment.

Progress on chronic studies and identifying no-effect concentrations

Robust PNEC or water quality guideline values for NMs in aquatic ecosystems are still not available to any great extent. Quality-assured chronic toxicity data are ideally required for PNECs, with ecologically relevant endpoints based on lethality, immobilization, growth, development, and reproduction, expressed preferably as EC/IC10 values, rather than no-observed-effect concentrations (NOECs). The lack of chronic toxicity data is problematic and a key research gap.

Chronic effects of NMs on reproduction have been studied in invertebrates, but the data so far are mostly the OECD-style chronic tests in freshwater using model organisms such as algae or cladocerans. These limited studies do show effects on growth and reproductive success (TiO₂ and ZnO NMs: Wiench et al.

2009; Ag NMs: Zhao and Wang 2011), with chronic reproductive endpoints being more sensitive than acute mortality. Developmental defects are reported for planktonic stages of sea squirts, marine mussels, and sea urchins with Fe particles (Kadar et al. 2013). Fullerenes are also known to interfere with the development of oysters, *Crassostrea virginica* (Ringwood et al. 2009).

Chronic effects on reproduction and offspring development have been studied to some extent in fish or amphibian models. The literature on amphibians has mainly focused on developmental toxicity to the growing tadpoles and their subsequent morphogenesis into the body form of adult frogs. Similar to fish embryos (Shaw et al. 2016), exposure of frog embryos to NMs prior to hatching has limited effects in terms of acute mortality, but at very high concentrations there can be some subsequent developmental defects (e.g., exposures to 1000 mg/L of TiO₂, Fe₂O₃, CuO, and ZnO NMs: Nations et al. 2011). Clearly, in unhatched embryos of both fishes and amphibians, the chorion and the mucilage of the perivitelline matrix is a formidable barrier to exposure. Animals therefore become more vulnerable when they have hatched. Early studies on *Xenopus laevis* tadpoles found that double-walled CNTs compacted the gills and gut to impair growth (Mouchet et al. 2008). There are also concerns that co-exposure of frogs to UV light and TiO₂ NMs may enhance ecotoxicity (Zhang J et al. 2012). However, the science is still at the observational and hypothesis-formulation stage for amphibians. There are only a few mechanistic studies, and to our knowledge only one of these is near environmentally relevant concentrations. Carew et al. (2015) used low µg/L concentrations of Ag NMs to investigate chronic effects on amphibian development. Delayed and abnormal development occurred. Critically, they identified interference with the thyroid hormone signaling that controls development as a key mechanism. More studies of this kind are needed on vertebrate and invertebrate animals. The usual assumption in ecotoxicology is that hazardous chemicals delay or adversely alter developmental anatomy. However, this does not have to be the case. One recent report showed that chronic exposure to Au NMs in the wood frog (*Lithobates sylvaticus*) can accelerate amphibian development rather than slow it (Fong et al. 2016).

Mesocosm studies

As with traditional chemicals, acute laboratory tests offer little environmental realism, and mesocosms provide potentially useful platforms for generating more realistic data. This realism is likely more important for NMs, given the system complexity, and it can be more easily generated by non-traditional testing methods (Hjorth et al. 2017b), although replication and isolation of individual variables is challenging. In 2008, there were almost no data on mesocosms, but now some progress has been made. Mesocosm studies have been performed in freshwater and terrestrial systems (Lowry et al. 2012b; Colman et al. 2013). Interestingly, Bone et al. (2015) compared a freshwater mesocosm exposed to a range of Ag NMs with exposure in equivalent laboratory conditions in the same study. They found that the laboratory studies did not replicate the findings in the mesocosms and highlighted the complexity of organic carbon

processing and UV light penetration as potential explanations for the differences. In addition, mesocosm results (Colman et al. 2014) suggest that all forms of Ag (dissolved and different NM types) ultimately have similar effects, whereas experimental laboratory data suggest that a nano effect exists (Leclerc and Wilkinson 2014), and that Ag NP data are scattered (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript), depending on the nature of the NM, the organism, and the media. However, mesocosm dosing also affected outcomes with lower concentrations over time, resulting in more suspended NMs compared with single high doses (Baker et al. 2016), in agreement with concentration-dependent laboratory studies (Merrifield et al. 2017a). The agreement between mesocosm and laboratory data is encouraging where it happens, but further work is needed to rationalize data where they do not agree.

Freshwater mesocosm studies have shown that additions of Au NMs will cause total Au concentrations to increase in the sediments and/or associated surface biofilms (Ferry et al. 2009). These studies showed that elevated total metal concentrations may also be found in primary producers and aquatic invertebrates (Lowry et al. 2012b), but whether or not this represents a hazard by food-chain transfer or simply exposure due to particle settling/direct exposure is not clear. In other work, Bour et al. (2016a, 2016b) serially dosed a freshwater mesocosm with CeO₂ NMs over 4 wk and detected total Ce in the biofilms of primary producers (bacteria, fungi, etc.), the grazing chironomid larvae, and *Pleurodeles* larvae (amphibian tadpole) as the top predator in the experiment. However, the bioaccumulation pattern for Ce did not explain the observed toxicity in the mesocosm, and parallel experiments excluded trophic transfer/oral toxicity. Bour et al. (2016a, 2016b) concluded that a more complex set of processes was occurring in the mesocosm, perhaps with indirect toxicity due to alterations in the microbial community in the test system.

Steps toward assessment of environmental risk

Toxicity data, used in a species sensitivity distribution, allow the derivation of a PNEC as the concentration that is hazardous to a given percentage of species, typically 95% (HC5). Ideally, chronic effect data, usually EC/IC10 (10% effect) or NOEC values are required from at least 8 species representative of at least 4 taxonomic groups, but when many of the data are only acute median lethal concentration (LC)/EC50 values, these are typically converted to chronic NOECs using a default acute-to-chronic ratio of 10 (Warne et al. 2015), although 100 to 1000 may be used for NMs, given the uncertainty. A review of published toxicity data for NMs found that few data sets met these criteria (Batley et al. 2013). Deficiencies included reporting only acute toxicity data (LC50 values), reporting EC50 rather than EC10 values, and reporting nominal rather than measured concentrations, along with a lack of information on dosimetry, aspects of which have been discussed. Often, the tested NMs differed in their physicochemical properties and transformations were not monitored, making comparisons between studies difficult. There is a pressing need for multispecies toxicity data

for each of the common forms of each NM, especially for the most toxic form of each so that we can at least derive a conservative HC5 value for use in risk assessments.

Of the published PNEC value derivations (Adam et al. 2015; Garner et al. 2015; Semenzin et al. 2015), those of Garner et al. appear to be the most reliable, and the mean values calculated from the plots in that study are summarized in Table 1; however, even these rely heavily on converted acute data. For nano-TiO₂, Semenzin et al. (2015) derived an HC5 of 0.02 mg/L, much lower than the value in Table 1, but their value is very conservative because conversion of sensitive chronic algal IC50 data to NOECs used a factor of 100 (not 10), whereas conversion of acute LC/EC50 data to chronic NOECs used the same factor. For nano-CuO, Adam et al. (2015) derived only an acute HC5 of 0.15 mg Cu/L (0.19 mg CuO/L) above the 0.04 mg/L chronic HC5 value. Dissolution is a confounding factor with CuO and ZnO NMs and to a lesser extent Ag NMs, but not for CeO₂, TiO₂, and the carbon-based NMs. For uncoated Ag NMs, Batley (G.E. Batley, unpublished results) derived a value of 0.1 μg/L, which is lower than the Table 1 value. Much of this variability might be explained by different NM types or other parameters. Because of the large variability in observed toxicity as a function of particular coatings, and as a function of NM-specific surface area, particle-specific guideline values or PNECs are necessary. van Hoecke et al. (2009), for example, found that the chronic toxicity of nano-CeO₂ to reproduction of the cladoceran *D. magna* decreased by a factor of 2 as the nominal particle size increased from 12 to 29 nm (hydrodynamic diameter range of 479–552 nm), as shown in Figure 7.

The simplest screening-level hazard assessment requires that the PEC/PNEC ratio should not exceed 1. In the cases just described, the modeled and measured environmental concentrations (PECs) in waters (Figure 2) were generally well below the derived PNEC values in Table 1, indicating a low risk to aquatic ecosystem health. It is anticipated that the same would be true for sediments. Nevertheless, given the uncertainties around issues such as dosimetry and chronic effects, and ongoing developments in NM sophistication and increased production levels, there is concern that this risk is severely underestimated and/or may increase in the near future.

A higher-level hazard assessment of chemicals would normally consider the persistence of the substance in the

environment and its potential for bioaccumulation and toxicity (i.e., persistence, bioaccumulation, and toxicity criteria). These issues also apply to NMs, but there are some additional considerations. First, the myriad of shapes, sizes, and chemical compositions of NMs precludes the evaluation of every single NM as a new substance in the testing strategy (Handy et al. 2012b). A more pragmatic approach is to group materials for hazard assessment and/or to estimate the hazard relative to existing substances of similar chemical composition (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript). Thus, in the present review we discuss whether or not patterns are emerging in the biological data that might allow this grouping and read-across to traditional chemicals. In terms of understanding the role of NM properties on a more fundamental level, clearly the production of tightly constrained NMs whose properties can be varied is essential (Baalousha and Lead 2013). This has not currently been performed to any great extent, but it is feasible. First, for example, the synthesis of NMs using enriched stable metal isotopes allows for traceability in bioaccumulation experiments at low concentrations (Dybowska et al. 2011; Larner et al. 2012; Croteau et al. 2014), whereas core-shell NMs provide a new tracing tool to characterize the mechanisms controlling bioavailability (Merrifield and Lead 2016). Secondly, a decade ago, there were many data gaps on hazard such that uncertainty was high, and thus systematic reconsideration of the data gaps is now warranted.

One difficulty with grouping hazard by chemical substance is how to classify composite materials, including NMs, that have a coating different from the core or core-shell NMs. Individual studies on aquatic species have shown coating-dependent toxicity and clearly these may be considered different materials, with the coating affecting many of the physicochemical and biological properties (see earlier section, *Selected Recent Advances in NM Composition and Metrology*). Coatings can also influence the dissolution of toxic metal ions from NMs (Zhao et al. 2012) and the exposure route (Croteau et al. 2011b). However, differentiating the indirect effects of NM behavior on toxicity from the inherent toxicity of the coating itself is challenging, and there are as yet no clear trends with respect to toxicity of surface coatings across a range of different organisms or NMs. In deciding how to treat these new materials, a key question relates to their persistence: are nanohybrids conserved over reasonable environmental timescales or do they rapidly dissociate into their component parts?

A final point is that, in the effort to establish the facts with laboratory testing, the overarching ecological principles of protecting most of the organisms most of the time, as well as preserving biodiversity and ecosystem function, may have been overlooked. The essence of environmental protection is to ensure the survival of the maximum number of species at the population level such that biodiversity and ecosystem functions remain intact. In freshwater systems, there are insufficient data to predict the prospects of long-term survival of most organisms. The situation is more uncertain for other compartments. The lack of knowledge regarding the effects of NMs on processes in river sediments and the biota at the base of aquatic food webs

TABLE 1: Mean chronic concentration that is hazardous to a given percentage of species, typically 95% (HC5) values for engineered nanoparticles in freshwaters based on species sensitivity distributions of chronic and converted acute data^a

Nanomaterial	HC5
Uncoated nano-Ag	1.2 μg/L
PVP-coated nano-Ag	0.7 μg/L
Nano-CuO	40 μg/L
Nano-CeO ₂	0.8 mg/L
Nano-TiO ₂	2 mg/L
CNTs	4.8 mg/L
Nano-C60	0.2 mg/L

^aFrom Garner et al. 2015.

PVP = polyvinylpyrrolidone; CNT = carbon nanotube.

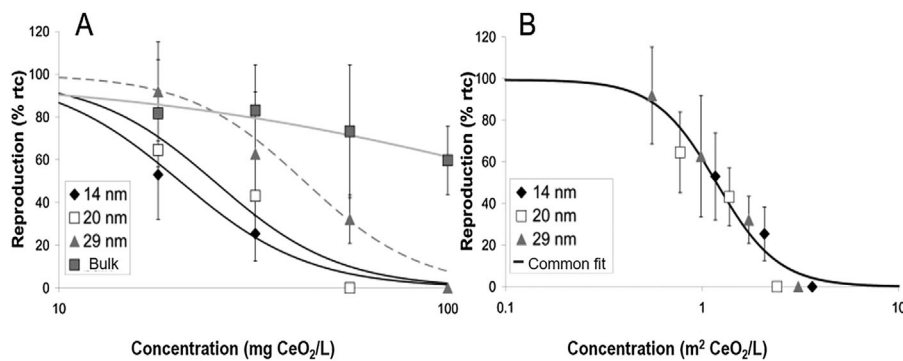


FIGURE 7: Concentration–response curves and calculated log–logistic fits in a 21-d reproduction test with *Daphnia magna* obtained for 14, 20, and 29 nm CeO₂ nanoparticles and CeO₂ bulk material. Concentration is expressed as mass (A) and as surface area (B). rtc = relative to control. (From van Hoecke et al. 2009.)

remains a barrier to understanding both the fate of NMs in ecosystems and their impact on functionality of biogeochemical processes such as the nitrogen cycle.

Regardless of the type of ecosystem or model organism, the ecotoxicity of organic NMs (other than CNTs) also remains poorly explored, as does the ecotoxicity of next-generation NMs. Progress on using quantitative structure–activity relationships (QSARs) for estimating uptake potential or ranking materials by hazard has been slow, mainly due to the lack of measurement methods to validate the QSAR models (Puzyn et al. 2011; Mu et al. 2016) with experimental data.

Hazard of nano- versus micron-sized materials in the aquatic environment

There were concerns in 2008 about novel materials generating previously unknown mechanisms of toxicity. It is now clear that there are nanospecific aspects to the bioavailability and toxicity of NMs. In vivo, significantly different biouptake rates (Croteau et al. 2011b, 2014; Khan et al. 2012, 2013a, 2013b; Thit et al. 2015) and molecular behaviors (Taylor et al. 2016) have been observed when the same element is in the nanoscale phase, compared with the dissolved or larger particle phase. The composition of an inorganic NM appears to be an important property determining bioavailability and toxicity (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript), possibly suggesting the importance of NM-related impacts on speciation/transformations and delivery to an organism. For example, SiO₂ and TiO₂ NMs are, in general, less toxic (Katsumiti et al. 2015; Lacave et al. 2016) than CuO, CdS, and CdSe NMs (Khan et al. 2013b; Katsumiti et al. 2014). Other properties, including those of the transformation products, are likely to be important in determining nanospecific effects (Römer et al. 2013).

This potential for nano-specific effects that would necessitate NMs being regulated differently with respect to their potential hazards to environmental health is a major concern for regulatory agencies; as discussed, there is clear evidence that there can be differences in the bioavailability and toxicity of NMs imparted by their smaller size. Concerns have been raised, for instance, about enhanced toxicity of traditional chemicals in the presence of

NMs, due to the so-called Trojan horse effect (see Baun et al. 2008; Ferreira et al. 2014).

In general, core composition drives relative hazard (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript). The acute toxicity to aquatic species has been reported in the $\mu\text{g/L}$ to mg/L range for a wide variety of NMs. At the upper end of this range, NMs might be considered a low acute hazard compared with chemicals not in the nanosized range. For example, certain metals such as Cd or Hg, and organics such as endosulfan, have higher acute toxicities in general. However, there are substantial limits to our understanding of transformations and dosimetry, which make the dose–response relationship of NMs much more complex than for these chemicals (Merrifield et al. 2017a).

Nonetheless, some common features are emerging. In many cases, for metal-containing NMs, the metal salt is more toxic than its equivalent nanoform (e.g., CuO NMs: Shaw et al. 2012) unless the particle dissolves and therefore shows similar toxicity to the free metal ion (ZnO NMs: Brun et al. 2014). However, there are a number of exceptions to this general rule. In some cases, toxicity can be attributed entirely to dissolution of the free metal ion (ZnO NMs: Franklin et al. 2007). Silver toxicity is often thought to be due to the Ag ion, although the data on this point are more equivocal, and a wide range of toxicities exist depending on test and NM properties. The most definitive demonstrations of the effect of nanosized (compared with micron-sized) particles can be seen when the confounding effects of solubility are minimized, as is the case with CeO₂ NMs (Batley et al. 2013), where distinctly greater toxicity of the nanoform was evident. This is best illustrated in Figure 8 by the chronic toxicity of ZnO, CeO₂, and Ag NMs to the sensitive microalga *P. subcapitata* in a synthetic soft water. Toxicity data for nano-TiO₂ and nano-CuO using the same alga have also been published (Aruoja et al. 2009). The IC₅₀ values for nano and bulk TiO₂ were, respectively, 5.8 and 35.9 $\text{mg TiO}_2/\text{L}$. Both forms aggregate, but the greater toxicity of the nanoform was largely due to coverage of the algal cells by nanoaggregates, whereas uncovered cells were present with bulk TiO₂. Due to their high specific surface area, NMs can also adsorb nutrients, limiting growth. The respective IC₅₀ values for ionic Cu²⁺ and nano and bulk CuO were 0.02, 0.71, and 11.6 mg/L (Aruoja et al.

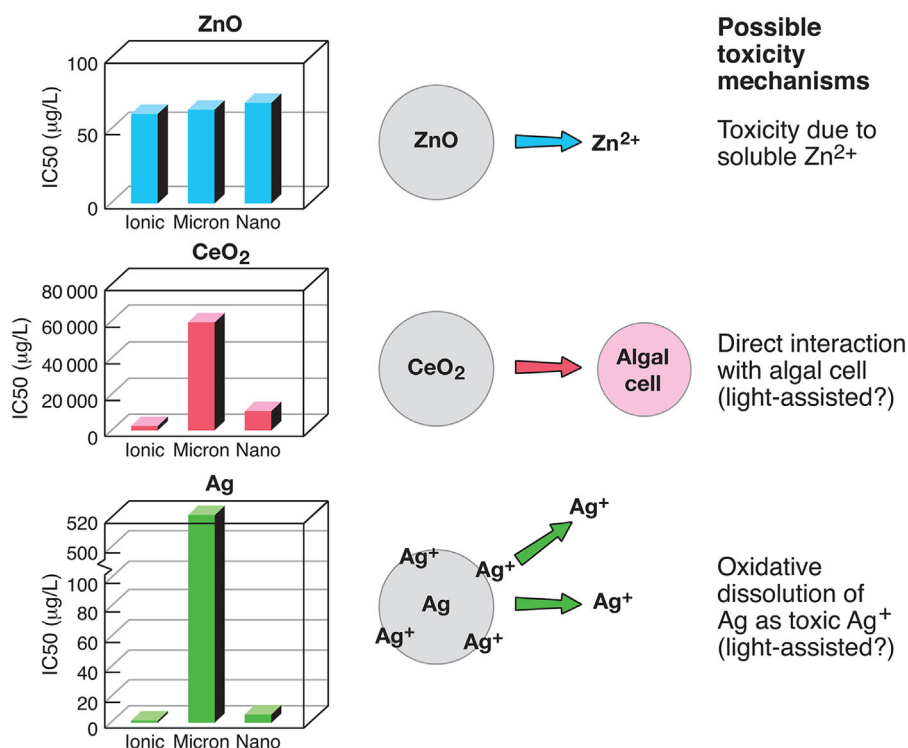


FIGURE 8: Comparison of the toxicities of micrometer-sized and nanosized ZnO, CeO₂, and Ag and their ionic equivalents Zn²⁺, Ce³⁺, and Ag⁺ to the freshwater alga *Pseudokirchneriella subcapitata* in a synthetic soft water. Toxicity values are IC₅₀s, the concentrations causing a 50% growth inhibition in µg/L. (Values for Zn from Franklin et al. 2007; values for Ce from Rogers et al. 2010; and values for Ag from Angel et al. 2013; reproduced from Batley et al. 2013 with permission.)

2009). Toxicity was largely explained by the greater concentration of soluble Cu released from the nanoform. These findings make a strong case for regulating metal NMs differently from the equivalent micron-sized particles, except in the extreme cases in which solubility nullifies the differences.

Although micron-sized materials are generally less toxic than their equivalent NM or the aggregated form (Römer et al. 2013), this is not always the case; pelagic and benthic organisms may be subject to different concentrations and forms of the NMs. Furthermore, the model system used to test toxicity can influence conclusions about relative toxicity. For example, in vivo models such as aquatic invertebrates appear less effective at differentiating toxicity than in vitro models such as human cell lines (S.N. Luoma, University of California, Davis, Davis, CA, USA, unpublished manuscript). Nanoscience has also created the opportunity to study some traditional chemicals in their bulk powder form, some of which have long been regarded as poorly soluble or of low/negligible toxicity to aquatic life. These studies have shown toxic effects from materials previously thought to be of a negligible hazard. For example, bulk forms of TiO₂ reduce embryo viability in the unexposed F1 generation offspring from exposed adult zebrafish (Ramsden et al. 2013). Moreover, the effect was similar to the nano form used in the experiment.

This discussion also raises a more general question related to controls. For instance, does the so-called free coating behave differently from the surface-bound coating? Are dissolved ion controls such as Ag suitable, because it is known that they precipitate and form new, smaller, and more bioavailable NMs

(Merrifield et al. 2017a)? The role of controls in nanotoxicology requires detailed, technical study and thought and is certainly not trivial.

BIOACCUMULATION AND TOXICITY OF NMs TO SOIL ORGANISMS

Bioaccumulation and toxicity of metal and metal oxide NMs in soils

Studies examining the toxicity and bioavailability of NMs to terrestrial organisms in soil-based media have increased in number over the past 8 to 10 y. However, studies of this nature remain uncommon, compared with studies examining either the aquatic ecotoxicology of NMs or the toxicity of NMs to terrestrial organisms in artificial exposure settings such as hydroponics. Early work in this area focused on as-manufactured NMs and includes reports of bioaccumulation of CuO and Ag NMs by earthworms (Unrine et al. 2010; Shoults-Wilson et al. 2011) and low bioavailability of CeO₂ NMs to corn (Birbauium et al. 2010), Fe₃O₄ NMs to pumpkin (Zhu et al. 2008), and TiO₂ and ZnO NMs to wheat (Du et al. 2011). More recently, studies have reported toxicity of FeO NMs to clover (Feng et al. 2013), alteration of nutrient localization in corn exposed to CeO₂ NMs (Zhao et al. 2015), and accumulation of CeO₂ NMs in soybean root nodules (Priester et al. 2012).

The strong interaction of most NMs with soils in terms of heteroaggregation with soil minerals and dissolved organic

matter raises doubts as to the environmental relevance of ecotoxicological thresholds developed using pristine NMs in non-soil media (Unrine et al. 2012a; Holden et al. 2016). Similarly, the effects of surface coatings, sizes, and shapes of pristine NMs on toxicity to soil organisms determined in non-soil media will also have little environmental relevance due to the reactions just described. More longer-term studies are needed to produce chronic toxicity data, because reactions of NMs in soils may take time to fully express (Diez-Ortiz et al. 2015).

Although early work with as-manufactured and relatively exotic NMs has provided useful information regarding their hazard, recent soil-based nanotoxicology research has concentrated on the NMs most likely to enter the terrestrial environment in large quantities and on the toxicity of NM transformation end products resulting from the chemical conditions associated with realistic discharge pathways. Consequently, this work has increasingly focused on TiO₂ and ZnO NMs because relatively high concentrations of these NMs are expected to be found in sludge (540 and 110 mg/kg 85th percentile concentrations in Europe for TiO₂ and ZnO, respectively; Sun et al. 2014); research has also been focused on Ag NMs because of their demonstrated biocidal properties and widespread use (Sondi and Salopek-Sondi 2004). Based on current NM usage patterns, TiO₂, ZnO, and Ag NMs and their transformation end products are expected to be introduced into terrestrial ecosystems in substantial amounts as a result of land application of biosolids. In recognition of this factor, exposures examining the toxicity of these NMs are increasingly taking place in biosolid-amended soil.

Sophisticated nanotoxicological studies are being undertaken in greater numbers, but nevertheless the potential risk posed to terrestrial ecosystems by metal and metal oxide NMs remains unclear for reasons similar to those for aquatic systems (limited dosimetry, transformation dynamics, and lack of chronic studies). Recent studies demonstrating transgenerational sensitivity in nematodes exposed to Ag and Ag₂S NMs (Schultz et al. 2016) and toxic effects in tomato progeny that were not observed in a parent generation exposed to CeO₂ NMs (Wang Q et al. 2013) highlight the potential importance of this type of research.

Judy et al. (2015b) reported data from a study examining the effects of amending soil with biosolids containing either bulk/dissolved TiO₂, Ag, and ZnO, nanoparticulate TiO₂, Ag, and ZnO, or containing no added metals (Chen et al. 2015; Judy et al. 2015b). These biosolids were generated using pilot WWTP facilities, and the metal input into the waste streams was subjected to chemical conditions similar to what they would experience in an actual WWTP facility. In this work, the NM treatment reduced root nodulation in the model legume *Medicago truncatula* relative to the bulk/dissolved control, a finding that was linked to significantly higher Zn bioaccumulation. Those studies also reported distinct shifts in the microbial communities in soil treated with NMs compared with the bulk/dissolved treatment. However, another study using the same media in a simpler system without plants reported that, although the microbial community in the bulk/dissolved treatment was significantly different from that in the NM treatment, this shift did

not result in a significant loss of ecosystem function (Duremkamp et al. 2016).

As with anoxic aquatic systems, a wide variety of Ag NMs will largely transform into insoluble Ag₂S NMs in a range of WWTP configurations (Lombi et al. 2013; Ma et al. 2013; Pradas del Real et al. 2016; Wang et al. 2016), although Ag-cysteine and AgCl are also possible transformation end products. Considering that non-labile transformation end products such as AgCl and Ag₂S are expected to be stable in the environment and to have relatively low bioavailability (Lombi et al. 2013; Donner et al. 2015; Doolette et al. 2015, 2016), the risk to terrestrial ecosystems posed by Ag NMs would appear to be low, although uncertainties related to longer-term transformations, dosimetry, are also relevant. This conclusion has been further supported by soil-based studies demonstrating that Ag₂S NMs presented minimal hazard to plant–microorganism symbioses (Judy et al. 2015a, 2016), crop plants (Doolette et al. 2015; Wang et al. 2016), soil microorganisms (Judy et al. 2015b; Doolette et al. 2016; Moore et al. 2016), and soil invertebrates (Starnes et al. 2015, 2016).

Du et al. (2015) recently reported the effects of CeO₂ NMs (doses of 100 and 400 mg/kg) on wheat grown in the field for 7 mo. There were no effects on wheat biomass, grain yield, or bioaccumulation of Ce, but morphological and biochemical effects were observed, with reductions in chlorophyll, delayed flowering, and increased catalase and superoxide dismutase activities. More of these types of study are needed, in which interactions and transformations of NMs with soil components are captured and integrated over longer exposure periods.

Bioaccumulation and toxicity of carbon NMs in soils

Carbon-based NMs such as fullerenes and CNTs are also commonly used in consumer products. However, the masses of carbon NMs entering WWTPs are relatively low, and, unlike Ag NMs, evidence suggests that carbon NMs are only likely to be toxic at relatively high concentrations (Nyberg et al. 2008; Garner et al. 2015). Although some studies have reported toxicity at high concentrations in hydroponics (Canas et al. 2009; Hawthorne et al. 2012) and in vitro (Jia et al. 2005), the small amount of ecotoxicological work that has taken place in soil-based media has reported little toxicity to soil organisms (Nyberg et al. 2008; Ge et al. 2016). Such research includes studies reporting low bioavailability of CNTs to earthworms in both artificial and natural soils (Petersen et al. 2008a, 2008b, 2011), and studies indicating that fullerenes have no impact on soil microbial communities (Tong et al. 2007; Nyberg et al. 2008). Newer carbon-based NMs like graphene, for example, have to date received little study but also appear to have relatively low toxicity (Ge et al. 2016).

Toxicity of NMs to other terrestrial biota

There appear to be no published peer-reviewed studies on the ecotoxicity of NMs to reptiles or birds, and this is a concern given that developmental toxicity and disruption of endocrine

processes have been identified in amphibians (Mouchet et al. 2008; Nations et al. 2011). The concerns for mammalian wildlife remain. There are no studies on the bioaccumulation hazard to small mammals or similar wildlife, and no studies reporting toxicity in an ecological context. Skalska and Strużyńska (2015) reported neurotoxicity to mammals, but the findings were based on rat brain slice, in vivo injection, or cell culture studies at high (mg) doses that have no ecophysiological relevance.

An in vivo injection study (Hanini et al. 2016) reported abnormal body temperature and decreasing heart rate in rabbits exposed to $Zn_{0.8}Co_{0.2}Fe_2O_4$ NMs with organ pathology, but there were no metal salt controls or characterization of metal dissolution—the observations might simply be explained by metal toxicity rather than any particle effect. Nonetheless, the lack of information on small mammals that are not laboratory animals remains a critical data gap for environmental hazard assessment purposes.

Trophic transfer of NMs in terrestrial biota

Early research reported that Au NMs could biomagnify in a simulated tobacco (*Nicotiana tabacum*)–tobacco hornworm (*Manduca sexta*) caterpillar food chain (Judy et al. 2011), raising concerns regarding the potential for humans to be exposed to NMs via trophic transfer. This work was followed by other studies that reported trophic transfer of NMs in other simulated food chains including earthworm (*Eisenia fetida*) to bullfrog (*Rana catesbeiana*; Unrine et al. 2012b), zucchini (*Cucurbita pepo*) to cricket (*Acheta domesticus*; Hawthorne et al. 2014), and lettuce (*Lactuca sativa*) to cricket to darkling beetles (Tenebrionidea; De la Torre-Roche et al. 2015). Although each of these 3 studies reported trophic transfer, transferred amounts of NMs were small, and the biomagnification reported in the tomato-hornworm study was not observed. However, recently biomagnification of NMs was again reported in a terrestrial food chain, with CeO_2 NM accumulated in kidney bean (*Phaseolus vulgaris*) plants biomagnifying in Mexican bean beetles (*Epilachna varivestis*) as well as in consumers of the beetles, spined soldier bugs (*Podisus maculiventris*; Majumdar et al. 2016). Taken together, these studies indicate that trophic transfer of NMs is likely to 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 NMs remain unclear.

CELLULAR MECHANISMS OF TOXICITY

Klaire et al. (2008) identified concerns about the mechanisms of toxicity of NMs, which initially centered around oxidative damage to cellular components, such as proteins and nucleic acids, as well as the theoretical hazard of an interruption of energy (Nel et al. 2006, 2009). However, even though many studies have incorporated measures of redox cellular responses, interpretation of their role with respect to nanotoxicity remains difficult. On the one hand, redox responses are a part of a wide array of evolutionarily well-embedded cellular defence mechanisms

aiming to overcome perturbations and to promote organism survival and health (Jennings 2013). For example, Minghetti and Schirmer (2016) noted that fish gill cells in culture partly recovered over time from a negative impact on metabolic activity and lysosomal membrane integrity brought about by Ag NMs, with a concomitant induction of glutathione reductase mRNA. On the other hand, whether such cellular stress responses are a primary reaction of the cells to the NM, or are a secondary response because of other damage by NMs to the cells, remains mostly undecided. It has therefore become clear that it is important to take a more integrated view of cell barriers and responses of cells to shed light on the mechanisms of NMs on cells, the functional units of life (Schirmer 2014).

Cellular barriers

Epithelia and endothelia in animals. The integrity of cellular epithelial or endothelial barriers, such as the epithelia of lungs, gills, and the intestine, or the endothelia of blood vessels, is crucial for the gatekeeper function of these barriers. Epithelia act as selective environment–organism barriers to air, water, or food; they regulate the interaction of an organism with its surrounding environment. Endothelia, on the other hand, serve as selective internal permeability barriers. Finding NMs in different tissues of animals after exposure (see *NM Bioavailability and Bioaccumulation in Aquatic Organisms*) demonstrates the potential of NMs to overcome such barriers. Moreover, it is well known that NMs can disrupt epithelia, such as those of the intestine and gill (single-walled CNTs: Smith et al. 2007). Understanding the mechanisms underlying the passage or even damage to cellular barriers, however, is difficult to study in vivo, and thus, cell culture–based 2-compartment systems have been developed to study particle transport and toxicity.

In such a system, cells are cultured on a porous membrane, which separates an upper (apical) and a lower (basolateral) compartment, mimicking the 2 sides of the barrier in vivo (Schirmer 2014). Transport of particles can be followed by measurement of their translocation across the cell layer on the porous membrane. Toxicity to the cells is assessed either as a loss in the electrical resistance of the cell layer or as molecular/cellular changes impacting the normal functioning of cells. Studies applying such systems have revealed that NMs are able to use cellular energy-dependent transport mechanisms through the cells. Size, shape, and charge have been characterized to be determinants of this, although a clear relationship between particle characteristics and transport behavior in different epithelial models cannot yet be deduced (see Yacobi et al. 2008; Mahler et al. 2012). However, certain behaviors have been determined. Mahler et al. (2012) showed that 50-nm negatively charged polystyrene particles were passively transported via paracellular routes, whereas otherwise similar 200-nm particles followed a transcellular energy-dependent route.

To accomplish NM transport studies in cells from nonmammalian species, a recently established fish epithelial barrier system (Minghetti et al. 2017) based on a rainbow trout (*Oncorhynchus mykiss*) intestinal cell line has been used. The RTgutGC cell line (Kawano et al. 2011) was adopted to allow

study of particle translocation and toxicity in the intestine of fish (Geppert et al. 2016). The RTgutGC intestinal model forms a leaky epithelium, which is in accordance with the fish intestinal epithelium in vivo (Geppert et al. 2016; Minghetti et al. 2017). It nevertheless formed an effective barrier for polystyrene NMs (50 nm nominal size): after 24 h of exposure, approximately 80% of the particles remained in the apical compartment, and 9 to 16% were associated with the cells, leaving <10% permeation to the basolateral side. Permeation was temperature and time dependent, indicating that active transport mechanisms, such as endocytosis, might be at play (Geppert et al. 2016). This model system thus offers a wealth of opportunities to study NM-affected barrier function in fish. The role of transformations before, during, and after transfer needs to be addressed in parallel with such a system.

Cell-based investigations that allow mechanisms to be explored under well-defined conditions and in a variety of processes (such as kinetics of transport in difficult to observe or to obtain tissue) are the only ways to derive experimental validation of hypotheses. However, such studies are limited because certain interactions and processes in and between tissues are difficult to replicate in vitro. One example with regard to an environment–organism barrier is described by Millaku et al. (2013). They studied the interaction of tungsten nanofibers in vivo in the model invertebrate terrestrial isopod *Porcellio scaber*. Hepatopancreatic cells of this organism were directly exposed to substances in partly digested food and filtered and transported from the stomach into the lumen of the hepatopancreas. In this process, the digestive gland epithelium is subject to physical forces that ensure peristaltic movement. They therefore proposed that muscle contraction during peristalsis may contribute to fiber insertion into cells and impact on the structure of the lining of the digestive tract. Similarly, Bacchetta et al. (2014) observed ZnO NMs in frog embryo enterocytes. The NMs were attached to the apical cell membrane or internalized into microvilli. An interaction between NMs and enterocytes led to the induction of oxidative stress and altered structural and functional integrity of the junctions between the cells.

Cell wall as barrier in microorganisms and plants. The cell wall can be envisioned to be an efficient barrier to combat penetration of NMs into cell wall-bearing cells. However, there are few mechanistic studies that attempt to understand the extent to which cell walls provide a protective function, or which structures and functions are protective. Navarro et al. (2008) postulated a sieve function of the cell wall, considering that pores across the cell wall are 5 to 20 nm in size. With improved methodologies, an increasing number of studies show unequivocal uptake of NMs into plants. After growth in soil spiked with CeO₂ NMs, CeO₂ NMs were discovered in soybean roots and root nodules (Priester et al. 2012). In addition, Zhao et al. (2015) found transfer of CeO₂ from soil into roots and shoots of corn plants with higher transfer into shoots in a carbon-poor compared with a carbon-rich soil. The presence of CeO₂ NMs around vascular vessels suggested that the particles found their way to the transport system and moved through the xylem driven by transpiration. In their study with CNTs, Miralles et al. (2012)

found adsorption of the tubes to alfalfa and wheat without significant uptake or translocation. More systematic studies are required to resolve the extent to which plant cell walls act as a barrier and for which NMs. Focusing on the interaction of Ag NMs with unicellular algae, Piccapietra et al. (2012) and Li et al. (2015) confirmed a strong interaction of particles with the cell surface of *Chlamydomonas reinhardtii* and *Euglena gracilis*, respectively, but with negligible uptake. The comparatively thick cell wall of algae appears to be a significant barrier to NM uptake.

These findings are in stark contrast to the efficient uptake of the Ag NMs by fish cells (Yue et al. 2017). Thus, NMs may elicit toxicity to cell wall-containing organisms without having to actually enter the cells. One demonstration of this phenomenon was provided by Bondarenko et al. (2013) for bacteria and Ag NMs. Using 6 bacterial strains and carefully accounting for extracellular and intracellular Ag ion concentrations liberated from the Ag NMs, they showed that direct contact between the bacterial cells and the Ag NMs enhanced the toxicity of the Ag NMs. The mechanism of toxicity in the absence of internalization needs to be addressed. Clearly, more systematic investigations on the barrier function of the cell wall and the ability of plant, bacterial, and algal membranes to endocytose or otherwise interact with NMs are needed.

In a study of the mechanism of chronic toxicity of CeO₂ NMs to the microalga *P. subcapitata*, Angel et al. (2015) used hyperspectral imaging to demonstrate the coating of cells with NMs, potentially causing cell damage (Rogers et al. 2010). Coating was absent in the presence of dissolved organic carbon (humic acid), minimizing toxicity. Although CeO₂, like TiO₂, is a source of reactive oxygen species (ROS), the intensity of ROS production was found to be lower and not sufficient to cause oxidative damage (Angel et al. 2013). Previous toxicity studies have demonstrated the role of ROS in the toxicity of TiO₂ to a range of species including algae, cladocerans, and fish (Ma et al. 2012; Miller et al. 2012).

Extracellular matrices. Many types of cells secrete molecules that form an extracellular matrix in the direct vicinity (the unstirred layer in Figure 6) to protect the cells and for specific interactions of cells with their immediate environment. Mucus-secreting cells in the external epithelia of organisms provide a layer of mucus that can temporarily protect the exterior. Mucus is a heterogeneous aqueous mixture of glycoproteins, among a number of other molecules, and its composition is highly conserved across biological species (for a review, see Handy and Maunder 2009). Similarly, bacteria, fungi, and/or algae forming biofilms in freshwater or soil, synthesize a matrix of extracellular polymeric substances. This matrix contains enzymes that hydrolyze dissolved high-molecular-weight compounds into small biomolecules, and only these can subsequently be taken up by microorganisms. Thus, the extracellular matrix has important physiological roles, which need to be considered when one is elucidating the mechanisms of toxicity of NMs.

To determine whether NMs can stimulate mucus secretion, Chen et al. (2011) exposed human bronchial epithelial cells to TiO₂ NMs and indeed found a concentration-dependent enhancement of mucin secretion (i.e., of glycoproteins forming the mucus). Mucin hypersecretion was strongly dependent on

the presence of intra- and extracellular Ca. Although the authors interpreted their finding as a link to the hypersecretion observed in chronic airway diseases, this example also demonstrates that a NM generally thought to be rather benign can elicit molecular changes with potential long-term consequences. Whether similar mechanisms of altered mucus secretion can be observed in cells of other organisms (fish, for example) would be an interesting route to pursue. Indeed, Smith et al. (2007) found increased gill mucus production after aqueous exposure of rainbow trout to single-walled CNTs.

A somewhat increased extracellular matrix production was observed after exposure to functionalized CdSe/ZnS quantum dots of the marine diatom *Thalassiosira pseudonana* (Zhang et al. 2013). Quantum dot agglomerates were associated with the extracellular matrix, which was thought to limit the toxicity of the quantum dots to the diatoms. This is an illustration of the importance of the behavior of NMs in the extracellular matrix to better account for NM bioavailability and the resulting protection or toxicity.

Extracellular enzymes of freshwater or soil biofilms provide essential nutrients to microorganisms and play an important role in nutrient cycling, but almost no knowledge about their susceptibility to NM exposure exists thus far. In an elaborate mesocosm study that explored the ecosystem effects of Ag NM exposure, Colman et al. (2013) found reduced levels of leucine aminopeptidase (responsible for nitrogen cycling) and alkaline phosphatase (phosphorus cycling) enzyme activity, which was closely linked to reduced microbial biomass. Exploring intact heterotrophic freshwater biofilms after exposure to differently functionalized TiO₂ NMs in the presence or absence of environmentally realistic UV radiation, Schug et al. (2014) found reduced activity of β -glucosidase (carbon cycling) and leucine aminopeptidase. Exploring other types of freshwater communities, Gil-Allué et al. (2015) and Tlili et al. (2016, 2017) demonstrated distinct impacts of Ag NMs on biochemical functions of the biofilms in periphyton and litter decomposers, respectively. For example, Tlili et al. (2016) found similar reductions in microbial respiration and utilization of carbon sources in biofilms exposed to Ag NMs and Ag nitrate.

Stress responses in cells

Once cell barriers are passed, distribution in the entire organism may ensue, and several response mechanisms can be expected to occur. Several recent reviews focus on mechanisms of NM cytotoxicity, intracellular targets, and signaling (see Marano et al. 2011; Fröhlich 2013). However, much of this discussion has focused on mammalian cells. Thus, there are large uncertainties about specific nonmammalian structures or functions, such as for algae, bacteria, and fish (Behra et al. 2013).

Cell membrane and cytoskeleton. Biophysical responses are likely to occur on contact of NMs with cell membranes (Wu et al. 2013). These processes include disturbance of the cell membrane's phospholipid bilayer based on the NM charge and the size-dependent formation of so-called holes (i.e., regions of reduced lipid or protein levels), which were shown to be

associated with cytotoxicity. Moreover, there were earlier reports on the ability of carbon-based NMs with diameters of approximately 1 nm to physically block membrane ion channels (Park et al. 2003). Once internalized, NMs may impair the cytoskeleton. For example, Soenen et al. (2010) proposed that high levels of Fe₂O₃ NMs, localized around the cell nucleus, hinder maturation of actin fibers due to steric interactions, which may severely affect cell migration and differentiation. More generally speaking, NMs can physically interact with biomolecules so that both biomolecule structure/function and NM behavior may change. Concepts of and experimental research (Shemetov et al. 2012) on these types of biophysical interactions are most advanced for peptides and proteins. For example, as particles get smaller and approach the size of some proteins, the curvature of the NM increasingly influences how proteins physically bind to the NM and thus change their proper folding. Active centers of enzymes or receptor binding sites might thereby become hidden (i.e., phasing the NM), or protein structure may be distorted in other ways and thus normal functioning can be impaired. These phenomena have thus far been studied almost exclusively in the context of NM-mediated drug delivery, imaging, or biosensing using human or other mammalian cells, and research has primarily focused on single-protein–NM interactions. It is important to perform these types of investigations with proteins or other biomolecules of particular relevance to organisms living in the environment, and with respect to potential long-term consequences of exposure. It has been shown by Linse et al. (2007), for example, that NM–protein interactions can lead to so-called amyloid (i.e., aggregated and deposited) proteins. Yue et al. (2016) developed a methodology to identify proteins that bind to Ag NMs in living fish gill cells. The list of candidate proteins obtained may guide such future work on specific NM–protein interactions.

Lysosomes. Another important consideration is the charge of the NM. Harush-Frenkel et al. (2008) demonstrated by means of a polylactic acid–based model NM that negatively charged particles preferentially localize in the lysosomes, whereas positively charged particles escape this route and are more predominantly found in the cytosol. The vast majority of environmentally relevant NMs tested thus far are negatively charged, and a variety of them have indeed been found colocalized with lysosomes (see Minghetti and Schirmer 2016; Yue et al. 2016). Accumulation in lysosomes may render organelles particularly susceptible to deleterious effects due to locally increased exposure. Yue et al. (2015) and Minghetti and Schirmer (2016) demonstrated that lysosomal membrane integrity in a fish gill and intestinal cell line was more susceptible to exposure to Ag NMs (but not to Ag ions) than metabolic activity or cell membrane integrity. Based on such findings, autophagy and lysosomal dysfunction have been proposed as a major route of NM toxicity (Stern et al. 2012). Autophagy is an evolutionarily conserved stress response for maintaining cellular homeostasis. Using Au NMs, Ma et al. (2011) demonstrated that NMs may cause autophagosome accumulation by interrupting the autophagic flux, specifically by impairing fusion with lysosomes and lysosome degradation capacity. Similar to

amyloidosis, defects in lysosome function may have serious long-term consequences for organism functioning (Stern et al. 2012). Moreover, in the short term, the acidic pH of lysosomes may affect the properties of NM, for instance through dissolution or desorption of metals. Xia et al. (2008), for example, demonstrated dissolution of ZnO NMs in lysosomes.

Mitochondria. With their central role in adenosine triphosphate production and apoptosis (i.e., programmed cell death), mitochondria are susceptible to NM exposure and, indeed, damage to mitochondria has been found after exposure of cells to different particles (see Fröhlich 2013). However, in contrast to lysosomes, mitochondria do not seem to be specifically targeted by NMs. Instead, their susceptibility appears to be most closely related to their propensity to cause oxidative stress, which can be caused by interference by NMs, or constituents thereof, with the mitochondrial electron transport chain. With regard to exposure of organisms in the environment, it would be particularly interesting to study whether NMs impact apoptosis as part of the normal development of many organ systems. Indeed, developmental malformations of the zebrafish embryo heart, which is the first organ to form and function in these embryos, were found after exposure to quantum dots (King-Heiden et al. 2009) and silica NMs (Duan et al. 2013). Whether these malformations were initiated by interference with normal apoptotic events was not specifically explored, but this factor appears to be particularly relevant with regard to potential delayed or long-term effects in organisms exposed to NMs during sensitive windows of development. Along these lines, Kim et al. (2013) determined that exposure of zebrafish embryos to Au NMs, which were surface-functionalized with a cationic ligand, caused smaller, malpigmented eyes. This defect was explained by a significant increase in cell death in the eye, which in turn correlated with an increase of the apoptosis-signaling genes p53 and bax. Disruption of proper eye development progressed into altered swimming behavior and reduced axon growth.

Nucleus. Translocation of NMs into the nucleus is, in principle, possible, although numerous studies, reporting on different non-dividing vertebrate cell lines and a variety of NMs in the ≥ 50 -nm range (Bastian et al. 2009; Kühnel et al. 2012; Busch et al. 2011; Yue et al. 2016), showed that NMs were detectable in the cytoplasm but not the nucleus. To allow bidirectional transport between the cytoplasm and the nucleus, the nuclear membrane contains specialized channels, called nuclear core complexes. As described by Paine et al. (1975) for the amphibian oocyte nuclear envelope, these complexes allow ions and small molecules to diffuse through aqueous channels with a diameter of approximately 9 nm. Accordingly, small particles, specifically quantum dots, have been shown to exploit the cells' active transport machinery and enter the nucleus via the nuclear pore complex (Nabiev et al. 2007). Molecules or particles larger than this, such as larger NMs or viruses, need to be selectively transported into the nucleus by tagging them with a signal (i.e., an amino acid tag called a nuclear localization sequence).

Panté and Kann (2002) demonstrated that Au NMs, coated with a nuclear protein and a receptor for its nuclear localization sequence, were able to move through the nuclear pore complex up to a total diameter of the particles (meaning including the protein coating) of approximately 39 nm. That study stresses the size- and signal-dependent translocation of molecules or particles into the nucleus under conditions in which cells do not divide. During cell division, however, nuclear envelope breakdown occurs, which allows mixing of cytoplasmic and nuclear compartments and transport of particles of approximately 100 nm (Lénárt et al. 2003). Indeed, Chen and von Mikecz (2005) reported on the translocation of up to 70-nm large silica particles into different, dividing mammalian cells. Translocation of the particles into the nucleus induced inhibition of gene expression and formation of protein aggregates, which resemble subnuclear pathologies also seen in neurodegenerative diseases and systemic autoimmune responses. Moreover, damage to DNA (genotoxicity) has been variously reported, for example, as micronuclei formation in mammalian cells by tungsten carbide (Kühnel et al. 2012) or Ag (Asharani et al. 2009) NMs. However, genotoxicity by itself cannot be taken as an indication that NMs have indeed entered the nucleus.

Among the mechanisms proposed for apparent NM-elicited genotoxicity are the production of ROS and mechanical interference. Both may be elicited without the NM having to enter the nucleus, for example, by oxidative stress and increased ROS production in the cytosol eventually leading to oxidative DNA damage, or by interference of NMs with actin filaments or microtubules during cell division (Gonzalez et al. 2008). In support of the theory of cytosolic ROS production as the initial insult to DNA damage, Wang et al. (2012) resolved DNA damage from CuO NM exposure of a human lung cell line to the following time-dependent series of events: oxidative stress associated with mitochondria occurring after 2 h of exposure, followed by enhanced transcript abundance of the stress response-sensitive protein kinases p38 and p53 at 4 h, and irreversible DNA damage starting to be measurable after 8 h of exposure. On the other hand, for quantum dots small enough to diffuse through the nuclear complex pore, targeting of histone proteins, which play an important role in cell cycle regulation and tumor growth, has been demonstrated (Nabiev et al. 2007). In summary, although damage of DNA by NM exposure has also been demonstrated in the absence of proven transport into nuclei, the mechanisms of impacting on DNA integrity and other functional or structural features of the nucleus, either directly or indirectly, warrant further investigations.

Caveat for sublethal exposure: Stimulatory hormetic effects

Hormesis is a biological response to a low dose of a toxicant whereby a beneficial effect may be observed (e.g., improved cell fitness, stress tolerance, growth, or longevity). This poorly understood phenomenon appears to be related to the activation of global cell repair mechanisms that overcompensate for the exposure to the toxicant (Calabrese 2001). Hormesis has been

shown to occur in human and mammalian cells exposed to NMs including nanotubes, quantum dots, ZnO, and Ag (Lavicoli et al. 2014). Moreover, bacterial exposure to sublethal concentrations of antimicrobial Ag NMs can induce various adaptation mechanisms that have a counterproductive hormetic effect on the intended microbial control, such as stimulated biofilm formation and increased antibiotic resistance (Yang and Alvarez 2015). Transcriptomic analysis revealed significant up-regulation of quorum sensing and lipopolysaccharide biosynthesis genes that enhance biofilm formation, possibly as a defense mechanism (Xiu et al. 2014). A hormetic effect (i.e., faster growth) was also observed for poplar trees and *Arabidopsis thaliana* (Wang J et al. 2013), which precludes generalizations about phytotoxicity. More discussion about hormetic effects can be found in the Supplemental Data.

CONCLUSIONS

The developments in the field since 2008 are large and important, and the emerging consensus is encouraging, with very pronounced developments in metrology, transformations, bioavailability, and mechanisms of toxicity. Nanoscience and nanotechnology are important social and economic drivers and will hopefully lead to improvements in society and well-being. The potential negative consequences to the environment have been relatively openly studied and discussed and this healthy transparent research environment has led to a general trust in the technology and its implementation. Much of the earlier research on the environmental implications of nanotechnology raised significant concerns based on reductionist experiments conducted under exposure conditions designed to elicit a response and discern toxicity mechanisms (e.g., unrealistically high doses, simple matrices). Recent experiments conducted under more realistic exposure conditions have found that the interactions of NMs with environmental matrices can, but do not necessarily, decrease their reactivity, bioavailability, and toxicity, which has suggested a more limited environmental risk in many cases. Despite the lack of major problems associated with NMs in the environment, there are very substantial data and knowledge gaps, including gaps in fundamental data on environmental concentrations. For instance, uncertainties in dosimetry remain, and although analytical advances are currently being made, these uncertainties make understanding the dose–response relationships challenging. In addition, the nanotechnology industry continues to grow, such that discharges and environmental burdens are rapidly increasing. Finally, nanotechnology innovation continues apace, with the development of more sophisticated and useful nano-enabled materials and products that are physicochemically and biologically active. A good example in the terrestrial environment is the increased use of NMs for pesticide and fertilizer release and delivery and the potential for nontargeted delivery of these and other chemicals. These products are often proprietary, and the nature of the NMs used in nanofertilizers and nanopesticides is often unclear, although applications employing Ag, CuO, and ZnO NMs are known. Trade secrecy related to proprietary materials again means that hazard and risk are much more

difficult to quantify accurately. These newer NMs are likely to be more persistent and stable, more dispersed, smaller, and multifunctional, which underscores the need to understand their potential unintended impacts. These uncertainties and rapid changes mean that there is a potential risk to the environment and to human health from environmental exposure that should not be ignored; greater integration of technology innovation and risk assessment is needed to ensure that the benefits of the technology are gained with a minimization of their possible negative effects. Continued scientific input is required to fill in our knowledge gaps and to judiciously apply the precautionary principle. The appropriate goal of these efforts is to help protect environmental and human health while ensuring the long-term sustainability of the nanotechnology industry.

Finally, and separate from a pragmatic assessment of risk, continued efforts to understand the environment and how we affect it are important scientific undertakings in their own right and can be justified solely on this basis. A better understanding of the science around NM fate, behavior, and effects is appropriate and should be undertaken even in the absence of any impact on regulation, risk assessment, and innovation.

Supplemental Data—The Supplemental Data are available on the Wiley Online Library at DOI: 10.1002/etc.4147.

Acknowledgment—J.R. Lead and G.E. Batley are joint lead coauthors. Partial funding was provided by the National Science Foundation (NSF CBET 1508931) and the Center for Environmental Nanoscience and Risk (grant number NSF CBET 1508931 to J. Lead), the NSF Engineering Research Center on Nanotechnology-Enabled Water Treatment (grant EEC-1449500 to P.J.J. Alvarez), the National Research Program and Toxics Substances Hydrology program of the US Geological Survey (to M-N. Croteau), and the European Union NanoFASE project (grant agreement no. 646002 to R.D. Handy). We thank S. Luoma and K. Wilkinson and 3 anonymous reviewers for their very helpful reviews. The section on mechanisms of toxicity was partially adapted and updated from the chapter by K. Schirmer in the 2014 book *Nanoscience and the Environment, Vol 7—Frontiers of Nanoscience* (Elsevier, Amsterdam, The Netherlands), which was coedited by J.R. Lead.

Disclaimer—Any use of trade, product, or firm names is for descriptive purposes only, and does not imply endorsement by the US Government.

Data Availability—Data are available in the Supplemental Data.

REFERENCES

- Adam V, Loyaux-Lawniczak S, Quaranta G. 2015. Characterization of engineered TiO₂ nanomaterials in a life cycle and risk assessments perspective. *Environ Sci Pollut Res* 22:11175–11192.
- Adeleye AS, Pokhrel S, Mädler L, Keller AA. 2018. Influence of nanoparticle doping on the colloidal stability and toxicity of copper oxide nanoparticles in synthetic and natural waters. *Water Res* 132:12–22.

- Aich N, Boateng LK, Sabaraya IV, Das D, Flora JRV, Saleh NB. 2016. Aggregation kinetics of higher-order fullerene clusters in aquatic systems. *Environ Sci Technol* 50:3562–3571.
- Aiken GR, Hsu-Kim H, Ryan JN. 2011. Influence of dissolved organic matter on the environmental fate of metals, nanoparticles, and colloids. *Environ Sci Technol* 45:3196–3201.
- Alabresm A, Mirshahghassemi S, Chandler GT, Decho AW, Lead JR. 2017. Use of PVP-coated magnetite nanoparticles to ameliorate oil toxicity to an estuarine meiobenthic copepod and stimulate the growth of oil degrading bacteria. *Environ Sci Nano* 4:1859–1865.
- Al-Bairuty GA, Shaw BJ, Handy RD, Henry TB. 2013. Histopathological effects of waterborne copper nanoparticles and copper sulphate on the organs of rainbow trout (*Oncorhynchus mykiss*). *Aquat Toxicol* 126:104–115.
- Al-Bairuty GA, Boyle D, Henry TB, Handy RD. 2016. Sublethal effects of copper sulphate compared to copper nanoparticles in rainbow trout (*Oncorhynchus mykiss*) at low pH: Physiology and metal accumulation. *Aquat Toxicol* 174:188–198.
- Al-Jubory AR, Handy RD. 2013. Uptake of titanium from TiO₂ nanoparticle exposure in the isolated perfused intestine of rainbow trout: Nystatin, vanadate and novel CO₂-sensitive components. *Nanotoxicology* 7:1282–1301.
- Angel BM, Batley GE, Jarolimek CV, Rogers NJ. 2013. The impact of size on the fate and toxicity of nanoparticulate silver in aquatic systems. *Chemosphere* 93:359–365.
- Angel BM, Vallotton P, Apte SC. 2015. On the mechanism of nanoparticulate CeO₂ toxicity to freshwater algae. *Aquat Toxicol* 168:90–97.
- Aruoja V, Dubourguier H-C, Kasemets K, Kahru A. 2009. Toxicity of nanoparticles of CuO, ZnO and TiO₂ to microalgae *Pseudokirchneriella subcapitata*. *Sci Total Environ* 407:1461–1468.
- Asharani PV, Mun GLK, Hande MP, Valiyaveetil S. 2009. Cytotoxicity and genotoxicity of silver nanoparticles in human cells. *ACS Nano* 3:279–290.
- Auvinen H, Kaegi R, Rousseau DPL, Du Laing G. 2017. Fate of silver nanoparticles in constructed wetlands—A microcosm study. *Water Air Soil Pollut* 228:97.
- Baalousha MA, Lead JR. 2013. Polydispersity in toxicology. *Nature Nanotech* 8:308–309.
- Baalousha M, Lead J. 2015. *Characterization of Nanomaterials in Complex Environmental and Biological Media*. Elsevier, Amsterdam, The Netherlands.
- Baalousha M, Manciualea A, Cumberland S, Kendall K, Lead JR. 2008. Aggregation and surface properties of iron oxide nanoparticles: Influence of pH and natural organic matter. *Environ Toxicol Chem* 27:1875–1882.
- Baalousha M, Ju-Nam Y, Cole PA, Gaiser B, Fernandes TF, Hriljac JA, Jepson MA, Stone V, Tyler CR, Lead JR. 2012. Characterization of cerium oxide nanoparticles—Part 1: Size measurements. *Environ Toxicol Chem* 31:983–993.
- Baalousha MA, Arkill KP, Romer I, Palmer RE, Lead JR. 2015. Transformations of citrate and Tween coated silver nanoparticles reacted with Na₂S. *Sci Total Environ* 502:344–353.
- Baalousha M, Cornelis G, Kuhlbusch T, Lynch I, Nickel C, Peijnenburg W, Van Den Brink N. 2016. Modeling nanomaterial fate and uptake in the environment: Current knowledge and future trends. *Environ Sci Nano* 3:323–345.
- Bacchetta R, Moschini E, Santo N, Fascio U, Del Giacco L, Freddi S, Camatini M, Mantecca P. 2014. Evidence and uptake routes for zinc oxide nanoparticles through the gastrointestinal barrier in *Xenopus laevis*. *Nanotoxicology* 8:728–744.
- Badawy AME, Luxton TP, Silva RG, Scheckel KG, Suidan MT, Tolaymat TM. 2010. Impact of environmental conditions (pH, ionic strength, and electrolyte type) on the surface charge and aggregation of silver nanoparticles suspensions. *Environ Sci Technol* 44:1260–1266.
- Baker LF, King RS, Unrine JM, Castellon BT, Lowry GV, Matson CW. 2016. Press or pulse exposures determine the environmental fate of cerium nanoparticles in stream mesocosms. *Environ Toxicol Chem* 35:1213–1223.
- Balasubramanian K, Burghard M. 2005. Chemically functionalized carbon nanotubes. *Small* 1:180–192.
- Bastian S, Busch W, Kühnel D, Springer A, Meißner T, Holke R, Scholz S, Iwe M, Pompe W, Gelinsky M. 2009. Toxicity of tungsten carbide and cobalt-doped tungsten carbide nanoparticles in mammalian cells *in vitro*. *Environ Health Perspect* 117:530.
- Batley GE, Kirby JK, McLaughlin MJ. 2013. Fate and risks of nanomaterials in aquatic and terrestrial environments. *Acc Chem Res* 46:854–862.
- Baun A, Sørensen SN, Rasmussen RF, Hartmann NB, Koch CB. 2008. Toxicity and bioaccumulation of xenobiotic organic compounds in the presence of aqueous suspensions of aggregates of nano-C₆₀. *Aquat Toxicol* 86:379–387.
- Beddow J, Stolpe B, Cole PA, Lead JR, Sapp M, Lyons BP, Colbeck I, Whitby C. 2017. Nanosilver inhibits nitrification and reduces ammonia-oxidising bacterial but not archaeal *amoA* gene abundance in estuarine sediments. *Environ Microbiol* 19:500–510.
- Behra R, Sigg L, Clift MJ, Herzog F, Minghetti M, Johnston B, Petri-Fink A, Rothen-Rutishauser B. 2013. Bioavailability of silver nanoparticles and ions: From a chemical and biochemical perspective. *J R Soc Interface* 10:20130396.
- Bi X, Lee S, Ranville JF, Sattigeri P, Spanias A, Herckes P, Westerhoff P. 2014. Quantitative resolution of nanoparticle sizes using single particle inductively coupled plasma mass spectrometry with the K-means clustering algorithm. *J Anal At Spectrom* 29:1630–1639.
- Bian S-W, Mudunkotuwa IA, Rupasinghe T, Grassian VH. 2011. Aggregation and dissolution of 4 nm ZnO nanoparticles in aqueous environments: Influence of pH, ionic strength, size, and adsorption of humic acid. *Langmuir* 27:6059–6068.
- Birbauium K, Brogioli R, Schellenberg M, Stark W, Gunther D, Limbach L. 2010. No evidence for cerium dioxide nanoparticle translocation in maize plants. *Environ Sci Technol* 44:8718–8723.
- Blickley TM, Matson CW, Vreeland WN, Rittschof D, Di Giulio RT, McClellan-Green PD. 2014. Dietary CdSe/ZnS quantum dot exposure in estuarine fish: Bioavailability, oxidative stress responses, reproduction, and maternal transfer. *Aquat Toxicol* 148:27–39.
- Bondarenko O, Ivask A, Kaminen A, Kurvet I, Kahru A. 2013. Particle-cell contact enhances antibacterial activity of silver nanoparticles. *PLoS One* 8:107–114.
- Bone AJ, Matson CW, Colman BP, Yang X, Meyer JN, Di Giulio RT. 2015. Silver nanoparticle toxicity to Atlantic killifish (*Fundulus heteroclitus*) and *Caenorhabditis elegans*: A comparison of mesocosm, microcosm, and conventional laboratory studies. *Environ Toxicol Chem* 34:275–282.
- Bour A, Mouchet F, Cadarsi S, Silvestre J, Chauvet E, Bonzom J-M, Pagnout C, Clivot H, Gauthier L, Pinelli E. 2016a. Impact of CeO₂ nanoparticles on the functions of freshwater ecosystems: A microcosm study. *Environ Sci Nano* 3:830–838.
- Bour A, Mouchet F, Cadarsi S, Silvestre J, Verneuil L, Baqué D, Chauvet E, Bonzom J-M, Pagnout C, Clivot H. 2016b. Toxicity of CeO₂ nanoparticles on a freshwater experimental trophic chain: A study in environmentally relevant conditions through the use of mesocosms. *Nanotoxicology* 10:245–255.
- Boxall A, Chaudhry Q, Sinclair C, Jones A, Aitken R, Jefferson B, Watts C. 2007. Current and future predicted environmental exposure to engineered nanoparticles. Central Science Laboratory, York, UK.
- Boyle D, Fox JE, Akerman JM, Sloman KA, Henry TB, Handy RD. 2014. Minimal effects of waterborne exposure to single-walled carbon nanotubes on behaviour and physiology of juvenile rainbow trout (*Oncorhynchus mykiss*). *Aquat Toxicol* 146:154–164.
- Bradford A, Handy RD, Readman JW, Atfield A, Muhling M. 2009. Impact of silver nanoparticle contamination on the genetic diversity of natural bacterial assemblages in estuarine sediments. *Environ Sci Technol* 43:4530–4536.
- Brown BE. 1982. The form and function of metal-containing granules in invertebrate tissues. *Biol Rev* 57:621–667.
- Brun NR, Lenz M, Wehrli B, Fent K. 2014. Comparative effects of zinc oxide nanoparticles and dissolved zinc on zebrafish embryos and eleutheroembryos: Importance of zinc ions. *Sci Total Environ* 476:657–666.
- Brunelli A, Pojana G, Callegaro S, Marcomini A. 2013. Agglomeration and sedimentation of titanium dioxide nanoparticles (n-TiO₂) in synthetic and real waters. *J Nanopart Res* 15:1684.
- Brunetti G, Donner E, Laera G, Sekine R, Scheckel KG, Khaksar M, Vasilev K, De Mastro G, Lombi E. 2015. Fate of zinc and silver engineered nanoparticles in sewerage networks. *Water Res* 77:72–84.
- Buffet P-E, Richard M, Caupos F, Vergnoux A, Perrein-Ettajani H, Luna-Acosta A, Akcha F, Amiard J-C, Amiard-Triquet C, Guibolini M. 2013. A mesocosm study of fate and effects of CuO nanoparticles on endobenthic species (*Scrobicularia plana*, *Hediste diversicolor*). *Environ Sci Technol* 47:1620–1628.

- Busch W, Bastian S, Trahorsch U, Iwe M, Kühnel D, Meißner T, Springer A, Gelinsky M, Richter V, Ikonomidou C. 2011. Internalisation of engineered nanoparticles into mammalian cells *in vitro*: Influence of cell type and particle properties. *J Nanopart Res* 13:293–310.
- Calabrese EJ. 2001. Overcompensation stimulation: A mechanism for hormetic effects. *Crit Rev Toxicol* 31:425–470.
- Canas JL, Long M, Nations S, Vadan R, Dai L, Luo, M, Ambikapathi R, Lee E, Olszyk D. 2009. Effects of functionalized and nonfunctionalized single-walled carbon nanotubes on root elongation of select crop species. *Environ Toxicol Chem* 27:1922–1931.
- Carew AC, Hoque ME, Metcalfe CD, Peyrot C, Wilkinson KJ, Helbing CC. 2015. Chronic sublethal exposure to silver nanoparticles disrupts thyroid hormone signaling during *Xenopus laevis* metamorphosis. *Aquat Toxicol* 159:99–108.
- Chaudhuri RG, Paria S. 2012. Core/shell nanoparticles: Classes, properties, synthesis mechanisms, characterization, and applications. *Chem Rev* 112:2373–2433.
- Chen M, von Mikecz A. 2005. Formation of nucleoplasmic protein aggregates impairs nuclear function in response to SiO₂ nanoparticles. *Experim Cell Res* 305:51–62.
- Chen EYT, Garnica M, Wang Y-C, Chen C-S, Chin W-C. 2011. Mucin secretion induced by titanium dioxide nanoparticles. *PLoS One* 6:1–8.
- Chen C, Unrine JM, Judy JD, Lewis RW, Guo J, McNear DH, Tsyusko OV. 2015. Toxicogenomic responses of the model legume *Medicago truncatula* to aged biosolids containing a mixture of nanomaterials (TiO₂, Ag, and ZnO) from a pilot wastewater treatment plant. *Environ Sci Technol* 49:8759–8768.
- Chouhan RS, Qureshi A, Yagci B, Gulgun MA, Ozguz V, Niazi JH. 2016. Biotransformation of multi-walled carbon nanotubes mediated by nanomaterial resistant soil bacteria. *Chem Eng J* 298:1–9.
- Colman BP, Wang SY, Auffan M, Wiesner MR, Bernhardt ES. 2012. Antimicrobial effects of commercial silver nanoparticles are attenuated in natural streamwater and sediment. *Ecotoxicology* 21:1867–1877.
- Colman BP, Arnaout CL, Anciaux S, Gunsch CK, Hochella MF, Kim B, Lowry GV, McGill BM, Reinsch BC, Richardson CJ, Unrine JM, Wright JP, Yin LY, Bernhardt ES. 2013. Low concentrations of silver nanoparticles in biosolids cause adverse ecosystem responses under realistic field scenario. *PLoS One* 8:10.
- Colman BP, Espinasse B, Richardson CJ, Matson CW, Lowry GV, Hunt DE, Wiesner MR, Bernhardt ES. 2014. Emerging contaminant or an old toxin in disguise? Silver nanoparticle impacts on ecosystems. *Environ Sci Technol* 48:5229–5236.
- Cornelis G, Kirby JK, Beak D, Chittleborough D, McLaughlin MJ. 2010. A method for determination of retention of silver and cerium oxide manufactured nanoparticles in soils. *Environ Chem* 7:298–308.
- Cornelis G, Ryan B, McLaughlin MJ, Kirby JK, Beak D, Chittleborough D. 2011. Solubility and batch retention of CeO₂ nanoparticles in soils. *Environ Sci Technol* 45:2777–2782.
- Cornelis G, Doolette C, Thomas M, McLaughlin MJ, Kirby JK, Beak D, Chittleborough D. 2012. Retention and dissolution of engineered silver nanoparticles in natural soils. *Soil Sci Soc Am J* 76:891–902.
- Cornelis G, Pang L, Doolette C, Kirby JK, McLaughlin MJ. 2013. Transport of silver nanoparticles in saturated columns of natural soils. *Sci Total Environ* 463:120–130.
- Cornelis G, Hund-Rinke K, Kuhlbusch T, Van den Brink N, Nickel C. 2014. Fate and bioavailability of engineered nanoparticles in soils: A review. *Crit Rev Environ Sci Technol* 44:2720–2764.
- Coutris C, Joner EJ, Oughton DH. 2012. Aging and soil organic matter content affect the fate of silver nanoparticles in soil. *Sci Total Environ* 420:327–333.
- Cozzari M, Elia AC, Pacini N, Smith BD, Boyle D, Rainbow PS, Khan FR. 2015. Bioaccumulation and oxidative stress responses measured in the estuarine ragworm (*Nereis diversicolor*) exposed to dissolved, nano- and bulk-sized silver. *Environ Pollut* 198:32–40.
- Cresswell T, Metian M, Golding LA, Wood MD. 2017. Aquatic live animal radiotracing studies for ecotoxicological applications: Addressing fundamental methodological deficiencies. *J Environ Radioact* 178–179:453–460.
- Cronholm P, Karlsson HL, Hedberg J, Lowe TA, Winnberg L, Elihn K, Wallinder IO, Moller L. 2013. Intracellular uptake and toxicity of Ag and CuO nanoparticles: A comparison between nanoparticles and their corresponding metal ions. *Small* 9:970–982.
- Croteau M-N, Dybowska AD, Luoma SN, Valsami-Jones E. 2011a. A novel approach reveals that zinc oxide nanoparticles are bioavailable and toxic after dietary exposures. *Nanotoxicology* 5:79–90.
- Croteau M-N, Misra SK, Luoma SN, Valsami-Jones E. 2011b. Silver bioaccumulation dynamics in a freshwater invertebrate after aqueous and dietary exposures to nanosized and ionic Ag. *Environ Sci Technol* 45:6600–6607.
- Croteau M-N, Misra SK, Luoma SN, Valsami-Jones E. 2014. Bioaccumulation and toxicity of CuO nanoparticles by a freshwater invertebrate after waterborne and dietborne exposures. *Environ Sci Technol* 48:10929–10937.
- Dai L, Banta GT, Selck H, Forbes VE. 2015. Influence of copper oxide nanoparticle form and shape on toxicity and bioaccumulation in the deposit feeder, *Capitella teleta*. *Mar Environ Res* 111:99–106.
- Dale AL, Casman EA, Lowry GV, Lead JR, Viparelli E, Baalousha M. 2015. Modeling nanomaterial environmental fate in aquatic systems. *Environ Sci Technol* 49:2587–2593.
- Darlington TK, Neigh AM, Spencer MT, Nguyen OT, Oldenburg SJ. 2009. Nanoparticle characteristics affecting environmental fate and transport through soil. *Environ Toxicol Chem* 28:1191–1199.
- De la Torre-Roche R, Servin A, Hawthorne J, Xing B, Newman LA, Ma X, Chen G, White JC. 2015. Terrestrial trophic transfer of bulk and nanoparticle La₂O₃ does not depend on particle size. *Environ Sci Technol* 49:11866–11874.
- Derfus AM, Chan WCW, Bhatia SN. 2004. Probing the cytotoxicity of semiconductor quantum dots. *Nano Lett* 4:11–18.
- Devi GP, Ahmed KBA, Varsha MKNS, Shrija BS, Lal KKS, Anbazhagan V, Thiagarajan R. 2015. Sulfidation of silver nanoparticle reduces its toxicity in zebrafish. *Aquat Toxicol* 158:149–156.
- Diegoli S, Manciualea AL, Begum S, Jones IP, Lead JR, Preece JA. 2008. Interaction between manufactured gold nanoparticles and naturally occurring organic macromolecules. *Sci Total Environ* 402:51–61.
- Diez-Ortiz M, Lahive E, George S, Ter Schure A, Van Gestel CAM, Jurkschat K, Svendsen C, Spurgeon DJ. 2015. Short-term soil bioassays may not reveal the full toxicity potential for nanomaterials; bioavailability and toxicity of silver ions (AgNO₃) and silver nanoparticles to earthworm *Eisenia fetida* in long-term aged soils. *Environ Pollut* 203:191–198.
- Domingos RF, Baalousha MA, Ju-Nam Y, Reid MM, Tufenkji N, Lead JR, Leppard GG, Wilkinson KJ. 2009. Characterizing manufactured nanoparticles in the environment: Multimethod determination of particle sizes. *Environ Sci Technol* 43:7277–7284.
- Donner E, Scheckel K, Sekine R, Popelka-Filcoff RS, Bennett JW, Brunetti G, Naidu R, McGrath SP, Lombi E. 2015. Non-labile silver species in biosolids remain stable throughout 50 years of weathering and aging. *Environ Pollut* 205:78–86.
- Doolette CL, McLaughlin MJ, Kirby JK, Navarro DA. 2015. Bioavailability of silver and silver sulfide nanoparticles to lettuce (*Lactuca sativa*): Effect of agricultural amendments on plant uptake. *J Hazard Mater* 300:788–795.
- Doolette CL, Vadakattu VSR, Gupta Lu Y, Payne JL, Batstone DJ, Kirby JK, Navarro DA, McLaughlin MJ. 2016. Quantifying the sensitivity of soil microbial communities to silver sulfide nanoparticles using metagenome sequencing. *PLoS One* 11:e0161979.
- Du W, Sun Y, Ji R, Zhu J, Wu J, Guo H. 2011. TiO₂ and ZnO nanoparticles negatively affect wheat growth and soil enzyme activities in agricultural soil. *J Environ Monit* 13:822–828.
- Du W, Gardea-Torresdey JL, Ji R, Yin Y, Zhu J, Peralta-Videa JR, Guo H. 2015. Physiological and biochemical changes imposed by CeO₂ nanoparticles on wheat: A life cycle field study. *Environ Sci Technol* 49:11884–11893.
- Duan J, Yu Y, Li Y, Yu Y, Sun Z. 2013. Cardiovascular toxicity evaluation of silica nanoparticles in endothelial cells and zebrafish model. *Biomaterials* 34:5853–5862.
- Dumont E, Johnson AC, Keller VDJ, Williams RJ. 2015. Nano silver and nano zinc-oxide in surface waters—Exposure estimation for Europe at high spatial and temporal resolution. *Environ Pollut* 196:341–349.
- Duremkamp M, Pawlett M, Ritz K, Harris JA, Neal AL, McGrath SP. 2016. Nanoparticles within WWTP sludges have minimal impact on leachate quality and soil microbial community structure and function. *Environ Pollut* 211:399–405.
- Dybowska AD, Croteau M-N, Misra SK, Berhanu D, Luoma SN, Christian P, O'Brien P, Valsami-Jones E. 2011. Synthesis of isotopically modified ZnO

- nanoparticles and their potential as nanotoxicity tracers. *Environ Pollut* 159:266–273.
- El Badawy AM, Scheckel KG, Suidan M, Tolaymat T. 2012. The impact of stabilization mechanism on the aggregation kinetics of silver nanoparticles. *Sci Total Environ* 429:325–337.
- Ellis LJA, Valsami-Jones E, Lead JR, Baalousha M. 2016. Impact of surface coating and environmental conditions on the fate and transport of silver nanoparticles in the aquatic environment. *Sci Total Environ* 568:95–106.
- Ellis LJA, Baalousha M, Valsami-Jones E, Lead JR. 2018. Seasonal variability of natural water chemistry affects the fate and behaviour of silver nanoparticles. *Chemosphere* 191:616–625.
- Fabrega J, Luoma SN, Tyler CR, Galloway TS, Lead JR. 2011. Silver nanoparticles: Behaviour and effects in the aquatic environment. *Environ Int* 37:517–531.
- Fabrega J, Zhang R, Renshaw J, Liu W-T, Lead JR. 2013. Impact of silver nanoparticles on natural marine biofilms. *Chemosphere* 85:961–966.
- Fang J, Shan X-Q, Wen B, Lin J-M, Owens G. 2009. Stability of titania nanoparticles in soil suspensions and transport in saturated homogeneous soil columns. *Environ Pollut* 157:1101–1109.
- Felix LC, Ortega VA, Goss GG. 2017. Cellular uptake and intracellular localization of poly (acrylic acid) nanoparticles in a rainbow trout (*Oncorhynchus mykiss*) gill epithelial cell line, RTgill-W1. *Aquat Toxicol* 192:58–68.
- Feng Y, Cui X, He S, Dong G, Chen M, Wang J, Lin X. 2013. The role of metal nanoparticles in influencing arbuscular mycorrhizal fungi effects on plant growth. *Environ Sci Technol* 47:9496–9504.
- Ferreira JLR, Lonné MN, Franca TA, Maximilla NR, Lugokenski TH, Costa PG, Fillmann G, Soares FAA, Fernando R, Monserrat JM. 2014. Co-exposure of the organic nanomaterial fullerene C₆₀ with benzo[a]pyrene in *Danio rerio* (zebrafish) hepatocytes: Evidence of toxicological interactions. *Aquat Toxicol* 147:76–83.
- Ferry JL, Craig P, Hexel C, Sisco P, Frey R, Pennington PL, Fulton MH, Scott GI, Decho AW, Kashiwada S, Murphy CJ, Shaw TJ. 2009. Transfer of gold nanoparticles from the water column to the estuarine food web. *Nat Nanotechnol* 4:441–444.
- Fong PP, Thompson LB, Carfagno GF, Sittan AJ. 2016. Long-term exposure to gold nanoparticles accelerates larval metamorphosis without affecting mass in wood frogs (*Lithobates sylvaticus*) at environmentally relevant concentrations. *Environ Toxicol Chem* 35:2304–2310.
- Franklin NM, Rogers NJ, Apte SC, Batley GE, Gadd GE, Casey PS. 2007. Comparative toxicity of nanoparticulate ZnO, bulk ZnO, and ZnCl₂ to a freshwater microalga (*Pseudokirchneriella subcapitata*): The importance of particle solubility. *Environ Sci Technol* 41:8484–8490.
- Fröhlich E. 2013. Cellular targets and mechanisms in the cytotoxic action of non-biodegradable engineered nanoparticles. *Curr Drug Metab* 14:976–988.
- Galloway T, Lewis C, Dolciotti I, Johnston BD, Moger J, Regoli F. 2010. Sublethal toxicity of nano-titanium dioxide and carbon nanotubes in a sediment dwelling marine polychaete. *Environ Pollut* 158:1748–1755.
- García-Alonso J, Khan FR, Misra SK, Turmaine M, Smith BD, Rainbow PS, Luoma SN, Valsami-Jones E. 2011. Cellular internalization of silver nanoparticles in gut epithelia of the estuarine polychaete *Nereis diversicolor*. *Environ Sci Technol* 45:4630–4636.
- Garner K, Suh S, Lenihan HS, Keller AA. 2015. Species sensitivity distributions for engineered nanomaterials. *Environ Sci Technol* 49:5753–5759.
- Ge Y, Priester JH, Mortimer M, Chang CH, Ji Z, Schimmel JP, Holden PA. 2016. Long-term effects of multiwalled carbon nanotubes and graphene on microbial communities in dry soil. *Environ Sci Technol* 50:3965–3974.
- Geppert M, Sigg L, Schirmer K. 2016. A novel two-compartment barrier model for investigating nanoparticle transport in fish intestinal epithelial cells. *Environ Sci Nano* 3:388–395.
- Gil-Allué C, Schirmer K, Tlili A, Gessner MO, Behra R. 2015. Silver nanoparticle effects on stream periphyton during short-term exposures. *Environ Sci Technol* 49:1165–1172.
- Gitrowski C, Al-Jubory AR, Handy RD. 2014. Uptake of different crystal structures of TiO₂ nanoparticles by Caco-2 intestinal cells. *Toxicol Lett* 226:264–276.
- Gomes T, Pereira CG, Cardoso C, Pinheiro JP, Cancio I, Bebianno MJ. 2011. Accumulation and toxicity of copper oxide nanoparticles in the digestive gland of *Mytilus galloprovincialis*. *Aquat Toxicol* 118–119:72–79.
- Gonzalez L, Lison D, Kirsch-Volders M. 2008. Genotoxicity of nanomaterials: A critical review. *Nanotoxicology* 2:252–273.
- Gottschalk F, Nowack B. 2012. Modeling the environmental release and exposure of engineered nanomaterials. In Puzyn T, Leszczynski J, eds, *Towards Efficient Designing of Safe Nanomaterials: Innovative Merge of Computational Approaches and Experimental Techniques*, RSC Nanoscience and Nanotechnology No. 25. The Royal Society of Chemistry, London UK, pp 284–313.
- Gottschalk F, Sonderer T, Scholz RW, Nowack B. 2009. Modeled environmental concentrations of engineered nanomaterials (TiO₂, ZnO, Ag, CNT, fullerenes) for different regions. *Environ Sci Technol* 43:9216–9222.
- Gottschalk F, Sonderer T, Scholz RW, Nowack B. 2010. Possibilities and limitations of modeling environmental exposure to engineered nanomaterials by probabilistic material flow analysis. *Environ Toxicol Chem* 29:1036–1048.
- Gottschalk F, Ort C, Scholz RW, Nowack B. 2011. Engineered nanomaterials in rivers—Exposure scenarios for Switzerland at high spatial and temporal resolution. *Environ Pollut* 159:3439–3445.
- Gottschalk F, Sun TY, Nowack B. 2013. Environmental concentrations of engineered nanomaterials: Review of modeling and analytical studies. *Environ Pollut* 181:287–300.
- Gray EP, Coleman JG, Bednar AJ, Kennedy AJ, Ranville JF, Higgins CP. 2013. Extraction and analysis of silver and gold nanoparticles from biological tissues using single particle inductively coupled plasma mass spectrometry. *Environ Sci Technol* 47:14315–14323.
- Grillet N, Manchon D, Cottancin E, Bertorelle F, Bonnet C, Broyer M, Lermé J, Pellarin M. 2013. Photo-oxidation of individual silver nanoparticles: A real-time tracking of optical and morphological changes. *J Phys Chem C* 117:2274–2282.
- Hadioui M, Leclerc S, Wilkinson KJ. 2013. Multimethod quantification of Ag⁺ release from nanosilver. *Talanta* 105:15–19.
- Hadioui M, Merdzan V, Wilkinson KJ. 2015. Detection and characterization of ZnO nanoparticles in surface and waste waters using single particle ICPMS. *Environ Sci Technol* 49:6141–6148.
- Handy RD, Maunder RJ. 2009. The biological roles of mucus: Importance for osmoregulation and osmoregulatory disorders of fish health. In Handy RA, Bury N, Flik, eds, *Osmoregulation and Ion Transport: Integrating Physiological, Molecular and Environmental Aspects*, Vol 1—Essential Reviews in Experimental Biology. Society for Experimental Biology, London, pp 203–235.
- Handy RD, Kammer FVD, Lead JR, Hassellöv M, Owen R, Crane M. 2008a. The ecotoxicology and chemistry of manufactured nanoparticles. *Ecotoxicology* 17:287–314.
- Handy RD, Henry TB, Scown TM, Johnston BD, Tyler CR. 2008b. Manufactured nanoparticles: Their uptake and effects on fish—A mechanistic analysis. *Ecotoxicology* 17:396–409.
- Handy RD, Al-Bairuty G, Al-Jubory A, Ramsden CS, Boyle D, Shaw BJ, Henry TB. 2011. Effects of manufactured nanomaterials on fishes: A target organ and body systems physiology approach. *J Fish Biol* 79:821–853.
- Handy RD, Brink NVD, Chappell M, Mühlhng M, Behra R, Dušinská M, Simpson P, Ahtainen J, Jha AN, Seiter J. 2012a. Practical considerations for conducting ecotoxicity test methods with manufactured nanomaterials: What have we learnt so far? *Ecotoxicology* 21:933–972.
- Handy RD, Cornelis G, Fernandes T, Tsyusko O, Decho A, Sabo-Attwood T, Metcalfe C, Steevens JA, Klaine SJ, Koelmans AA. 2012b. Ecotoxicity test methods for engineered nanomaterials: Practical experiences and recommendations from the bench. *Environ Toxicol Chem* 31:15–31.
- Hansen SF, Hjorth R, Skjolding LM, Bowman DM, Maynard A, Baun A. 2017. A critical analysis of the environmental dossiers from the OECD sponsorship programme for the testing of manufactured nanomaterials. *Environ Sci Nano* 4:282–291.
- Hartmann NB, Ågerstrand M, Lützhøft HCH, Baun A. 2017. NanoCred: A transparent framework to assess the regulatory adequacy of ecotoxicity data for nanomaterials—Relevance and reliability revisited. *NanoImpact* 6:81–89.
- Hartmann NIB, Skjolding LM, Hansen SF, Baun A, Kjølholt J, Gottschalk F. 2014. Environmental fate and behaviour of nanomaterials: New knowledge on important transformation processes. Danish Environmental Protection Agency, Copenhagen, Denmark.

- Harush-Frenkel O, Rozentur E, Benita S, Altschuler Y. 2008. Surface charge of nanoparticles determines their endocytotic and transcytotic pathway in polarized MDCK cells. *Biomacromolecules* 9:435–443.
- Hassellöv M, Kaegi R. 2009. *Analysis and Characterization of Manufactured Nanoparticles in Aquatic Environments*. John Wiley & Sons, Hoboken, NJ, USA.
- Hawthorne J, Musante C, Sinha SK, White JC. 2012. Accumulation and phytotoxicity of engineered nanoparticles to *Cucurbita pepo*. *Int J Phytoremed* 14:429–442.
- Hawthorne J, De la Torre Roche R, Xing B, Newman LA, Ma X, Majumdar S, Gardea-Torresdey J, White JC. 2014. Particle-size dependent accumulation and trophic transfer of cerium oxide through a terrestrial food chain. *Environ Sci Technol* 48:13102–13109.
- Hanini A, El Massoudi M, Gavard J, Kacem K, Ammar S, Souilem O. 2016. Nanotoxicological study of polyol-made cobalt-zinc ferrite nanoparticles in rabbit. *Environ Toxicol Pharmacol* 45:321–327.
- Heggelund LR, Diez-Ortiz M, Lofts S, Lahive E, Jurkschat K, Wojnarowicz J, Cedergreen N, Spurgeon D, Svendsen C. 2014. Soil pH effects on the comparative toxicity of dissolved zinc, non-nano and nano ZnO to the earthworm *Eisenia fetida*. *Nanotoxicology* 8:559–572.
- Heinlaan M, Kahru A, Kasemets K, Arbeille B, Prensier G, Dubourguier HC. 2011. Changes in *Daphnia magna* midgut after ingestion of copper oxide nanoparticles: A transmission electron microscopy study. *Water Res* 45:179–190.
- Hitchman A, Smith GHS, Ju-Nam Y, Sterling M, Lead JR. 2013. The effect of environmentally relevant conditions on PVP stabilised gold nanoparticles. *Chemosphere* 90:410–416.
- Hjorth R, Holden PA, Hansen SF, Colman BP, Grieger K, Hendren CO. 2017a. The role of alternative testing strategies in environmental risk assessment of engineered nanomaterials. *Environ Sci Nano* 4:292–301.
- Hjorth R, Skjolding LM, Sørensen SN, Baun A. 2017b. Regulatory adequacy of aquatic ecotoxicity testing of nanomaterials. *NanolImpact* 8:28–37.
- Holden PA, Gardea-Torresdey JL, Klaessig F, Turco RF, Mortimer M, Hund-Rinke K, Cohen Hubal EA, Avery D, Barceló D, Behra R, Cohen Y, Deydier-Stephan L, Ferguson PL, Fernandes TF, Herr Harthorn B, Henderson WM, Hoke RA, Hristozov D, Johnston JM, Kane AB, Kapustka L, Keller AA, Lenihan HS, Lovell W, Murphy CJ, Nisbet RM, Petersen EJ, Salinas ER, Scheringer M, Sharma M, Speed DE, Sultan Y, Westerhoff P, White JC, Wiesner MR, Wong EM, Xing B, Steele Horan M, Godwin HA, Nel AE. 2016. Considerations of environmentally relevant test conditions for improved evaluation of ecological hazards of engineered nanomaterials. *Environ Sci Technol* 50:6124–6145.
- Holmberg JP, Abbas Z, Ahlberg E, Hassellöv M, Bergenholtz J. 2011. Nonlinear concentration dependence of the collective diffusion coefficient of TiO₂ nanoparticle dispersions. *J Phys Chem C* 115:13609–13616.
- Honeyman BD, Santschi PH. 1991. Coupling adsorption and particle aggregation: Laboratory studies of “colloidal pumping” using iron-59-labeled hematite. *Environ Sci Technol* 25:1739–1747.
- Hoppe M, Mikutta R, Utermann J, Duijnsveld W, Kaufhold S, Stange CF, Guggenberger G. 2015. Remobilization of sterically stabilized silver nanoparticles from farmland soils determined by column leaching. *Eur J Soil Sci* 66:898–909.
- Hotze EM, Phenrat T, Lowry GV. 2010. Nanoparticle aggregation: Challenges to understanding transport and reactivity in the environment. *J Environ Qual* 39:1909–1924.
- Hou W-C, Westerhoff P, Posner JD. 2013. Biological accumulation of engineered nanomaterials: A review of current knowledge. *Environ Sci Process Impact* 15:103–122.
- Hughes GM, Al-Kadhomiy NK. 1988. Changes in scaling of respiratory systems during the development of fishes. *J Mar Biol Assoc UK* 68:489–498.
- Hull MS, Chaurand P, Rose J, Auffan M, Bottero JY, Jones JC, Schultz IR, Vikelsand PJ. 2011. Filter feeding bivalves store and deposit colloidal stable gold nanoparticles. *Environ Sci Technol* 45:6592–6599.
- Hund-Rinke K, Baun A, Cupi D, Fernandes TF, Handy R, Kinross JH, Navas JM, Peijnenburg W, Schlich K, Shaw BJ. 2016. Regulatory ecotoxicity testing of nanomaterials—Proposed modifications of OECD test guidelines based on laboratory experience with silver and titanium dioxide nanoparticles. *Nanotoxicology* 10:1442–1447.
- Huynh KA, McCaffery JM, Chen KL. 2012. Heteroaggregation of multiwalled carbon nanotubes and hematite nanoparticles: Rates and mechanisms. *Environ Sci Technol* 46:5912–5920.
- Jaisi DP, Elimelech M. 2009. Single-walled carbon nanotubes exhibit limited transport in soil columns. *Environ Sci Technol* 43:9161–9166.
- Jaisi DP, Saleh NB, Blake RE, Elimelech M. 2008. Transport of single-walled carbon nanotubes in porous media: Filtration mechanisms and reversibility. *Environ Sci Technol* 42:8317–8323.
- Jennings P. 2013. Stress response pathways, toxicity pathways and adverse outcome pathways. *Arch Toxicol* 87:13–14.
- Jia G, Wang H, Yan L, Wang X, Pei J, Yan T, Zhao Y, Guo X. 2005. Cytotoxicity of carbon nanomaterials: Single-wall nanotube, multi-wall nanotube, and fullerene. *Environ Sci Technol* 39:1378–1383.
- Johnston BD, Scown TM, Moger J, Cumberland SA, Baalousha M, Linge K, Aerle RV, Jarvis K, Lead JR, Tyler CR. 2010. Bioavailability of nanoscale metal oxides TiO₂, CeO₂, and ZnO to fish. *Environ Sci Technol* 44:1144–1151.
- Jovanović B, Palić D. 2012. Immunotoxicology of non-functionalized engineered nanoparticles in aquatic organisms with special emphasis on fish—Review of current knowledge, gap identification, and call for further research. *Aquat Toxicol* 118–119:141–151.
- Judy JD, Unrine JM, Bertsch PM. 2011. Evidence for biomagnification of gold nanoparticles within a terrestrial food chain. *Environ Sci Technol* 45:776–781.
- Judy JD, Kirby JK, Creamer C, McLaughlin MJ, Fiebiger C, Wright C, Cavagnaro T, Bertsch PM. 2015a. Effects of silver sulfide nanomaterials on mycorrhizal colonization of tomato plants and soil microbial communities in biosolid-amended soil. *Environ Pollut* 206:256–263.
- Judy JD, McNear D, Chen C, Lewis RW, Tsyusko OV, Bertsch PM, Rao W, Stegemeier J, Lowry GV, McGrath SP, Durenkamp M, Unrine JM. 2015b. Nanomaterials in biosolids inhibit nodulation, shift microbial community composition, and result in increased metal uptake relative to bulk metals. *Environ Sci Technol* 49:8751–8758.
- Judy JD, Kirby JK, McLaughlin MJ, McNear D, Bertsch PM. 2016. Symbiosis between nitrogen-fixing bacteria and *Medicago truncatula* is not significantly affected by silver and silver sulfide nanomaterials. *Environ Pollut* 214:731–736.
- Kadar E, Dyson O, Handy RD, Al-Subiai SN. 2013. Are reproduction impairments of free spawning marine invertebrates exposed to zero-valent nano-iron associated with dissolution of nanoparticles? *Nanotoxicology* 7:135–143.
- Kaegi R, Voegelin A, Sinnet B, Zuleeg S, Hagendorfer H, Burkhardt M, Siegrist H. 2011. Behavior of metallic silver nanoparticles in a pilot wastewater treatment plant. *Environ Sci Technol* 45:3902–3908.
- Kahru A, Dubourguier H-C. 2010. From ecotoxicology to nanoecotoxicology. *Toxicology* 269:105–119.
- Kasel D, Bradford SA, Simunek J, Putz T, Vereecken H, Klumpp E. 2013. Limited transport of functionalized multi-walled carbon nanotubes in two natural soils. *Environ Pollut* 180:152–158.
- Katsumita A, Gilliland D, Arostegui I, Cajaraville M. 2014. Cytotoxicity and cellular mechanisms involved in the toxicity of CdS quantum dots in hemocytes and gill cells of the mussel *Mytilus galloprovincialis*. *Aquat Toxicol* 153:39–52.
- Katsumita A, Berhanu D, Howard KT, Arostegui I, Oron M, Reip P, Valsami-Jones E, Cajaraville MP. 2015. Cytotoxicity of TiO₂ nanoparticles to mussel hemocytes and gill cells *in vitro*: Influence of synthesis method, crystalline structure, size and additive. *Nanotoxicology* 9:543–553.
- Kawano A, Haiduk C, Schirmer K, Hanner R, Lee Lej, Dixon B, Bols NC. 2011. Development of a rainbow trout intestinal epithelial cell line and its response to lipopolysaccharide. *Aquat Nutr* 17:E241–E252.
- Keller AA, Lazareva A. 2014. Predicted releases of engineered nanomaterials: From global to regional to local. *Environ Sci Technol Lett* 1:65–70.
- Keller AA, McFerran S, Lazareva A, Suh S. 2013. Global life cycle releases of engineered nanomaterials. *J Nanopart Res* 15:17.
- Khan FR, Misra SK, Garcá-Alonso J, Smith BD, Strekopytov S, Rainbow PS, Luoma SN, Valsami-Jones E. 2012. Bioaccumulation dynamics and modeling in an estuarine invertebrate following aqueous exposure to nanosized and dissolved silver. *Environ Sci Technol* 46:7621–7628.
- Khan FR, Laycock A, Dybowska A, Larner F, Smith BD, Rainbow PS, Luoma SN, Rehkämper M, Valsami-Jones E. 2013a. Stable isotope tracer to determine uptake and efflux dynamics of ZnO nano- and bulk particles and dissolved Zn to an estuarine snail. *Environ Sci Technol* 47:8532–8539.
- Khan FR, Schmuecking K, Krishnadasan SH, Berhanu D, Smith BD, deMello JC, Rainbow PS, Luoma SN, Valsami-Jones E. 2013b. Dietary

- bioavailability of cadmium presented to the gastropod *Peringia ulvae* as quantum dots and in ionic form. *Environ Sci Technol* 32:2621–2629.
- Khan FR, Kennaway GM, Croteau M-N, Dybowska A, Smith BD, Nogueira AJ, Rainbow PS, Luoma SN, Valsami-Jones E. 2014. *In vivo* retention of ingested Au NMs by *Daphnia magna*: No evidence for trans-epithelial alimentary uptake. *Chemosphere* 100:97–104.
- Khan FR, Paul KB, Dybowska AD, Valsami-Jones E, Lead JR, Stone V, Fernandes TF. 2015. Accumulation dynamics and acute toxicity of silver nanoparticles to *Daphnia magna* and *Lumbriculus variegatus*: Implications for metal modeling approaches. *Environ Sci Technol* 49:4389–4397.
- Kim KT, Zaikova T, Hutchison JE, Tanguay RL. 2013. Gold nanoparticles disrupt zebrafish eye development and pigmentation. *Toxicol Sci* 133:275–288.
- Kim B, Park CS, Murayama M, Hochella MF. 2010. Discovery and characterization of silver sulfide nanoparticles in final sewage sludge products. *Environ Sci Technol* 44:7509–7514.
- King-Heiden TC, Wicinski PN, Mangham AN, Metz KM, Nesbit D, Pedersen JA, Hamers RJ, Heideman W, Peterson RE. 2009. Quantum dot nanotoxicity assessment using the zebrafish embryo. *Environ Sci Technol* 43:1605–1611.
- Klaine SJ, Alvarez PJJ, Batley GE, Fernandes TF, Handy RD, Lyon DY, Mahendra S, McLaughlin MJ, Lead JR. 2008. Nanomaterials in the environment: Behavior, fate, bioavailability, and effects. *Environ Toxicol Chem* 27:1825–1851.
- Klaine SJ, Koelmans AA, Horne N, Carley S, Handy RD, Kapustka L, Nowack B, von der Kammer F. 2012. Paradigms to assess the environmental impact of manufactured nanomaterials. *Environ Toxicol Chem* 31:3–14.
- Klitzke S, Metreveli G, Peters A, Schaumann GE, Lang F. 2015. The fate of silver nanoparticles in soil solution—Sorption of solutes and aggregation. *Sci Total Environ* 535:54–60.
- Kühnel D, Scheffler K, Wellner P, Meissner T, Potthoff A, Busch W, Springer A, Schirmer K. 2012. Comparative evaluation of particle properties, formation of reactive oxygen species and genotoxic potential of tungsten carbide based nanoparticles in vitro. *J Hazard Mater* 227:418–426.
- Labille J, Harns C, Bottero JY, Brant J. 2015. Heteroaggregation of titanium dioxide nanoparticles with natural clay colloids. *Environ Sci Technol* 49:6608–6616.
- Lacave JM, Retuerto A, Vicario-Parés U, Gilliland D, Oron M, Cajaraville MP, Orbea A. 2016. Effects of metal-bearing nanoparticles (Ag, Au, CdS, ZnO, SiO₂) on developing zebrafish embryos. *Nanotechnology* 27:325102.
- Lanno RP, Hicks B, Hilton JW. 1987. Histological observations on intrahepatic copper-containing granules in rainbow trout reared on diets containing elevated levels of copper. *Aquat Toxicol* 10:251–263.
- Larner F, Dogra Y, Dybowska A, Fabrega J, Stolpe BR, Bridgestock LJ, Goodhead R, Weiss DJ, Moger J, Lead JR. 2012. Tracing bioavailability of ZnO nanoparticles using stable isotope labeling. *Environ Sci Technol* 46:12137–12145.
- Laurent P, Hebibi N. 1989. Gill morphometry and fish osmoregulation. *Can J Zool* 67:3055–3063.
- Laux P, Riebeling C, Booth AM, Brain JD, Brunner J, Cerrillo C, Creutzenberg O, Estrela-Lopis I, Gebel T, Johanson G, Jungnickel H, Kock H, Tentschert J, Tlili A, Schäffer A, Sips AJAM, Yokel RA, Luch A. 2018. Challenges in characterizing the environmental fate and effects of carbon nanotubes and inorganic nanomaterials in aquatic systems. *Environ Sci Nano* 5: 48–63.
- Lavicoli I, Fontana L, Leso V, Calabrese EJ. 2014. Hormetic dose-responses in nanotechnology studies. *Sci Total Environ* 487:361–374.
- Lead JR, Muirhead D, Gibson CT. 2005. Characterization of freshwater natural aquatic colloids by atomic force microscopy (AFM). *Environ Sci Technol* 39:6930–6936.
- Leclerc S, Wilkinson KJ. 2014. Bioaccumulation of nanosilver by *Chlamydomonas reinhardtii*—Nanoparticle or the free ion? *Environ Sci Technol* 48: 358–364.
- Lecoanet HF, Bottero J-Y, Wiesner MR. 2004. Laboratory assessment of the mobility of nanomaterials in porous media. *Environ Sci Technol* 38: 5164–5169.
- Lee S, Bi X, Reed RB, Ranville JF, Herckes P, Westerhoff P. 2014. Nanoparticle size detection limits by single particle ICP-MS for 40 elements. *Environ Sci Technol* 48:10291–10300.
- Lénárt P, Rabut G, Daigle N, Hand AR, Terasaki M, Ellenberg J. 2003. Nuclear envelope breakdown in starfish oocytes proceeds by partial NPC disassembly followed by a rapidly spreading fenestration of nuclear membranes. *J Cell Biol* 160:1055–1068.
- Lesniak W, Bielinska AU, Sun K, Janczak KW, Shi X, Baker JR, Balogh LP. 2005. Silver/dendrimer nanocomposites as biomarkers: Fabrication, characterization, in vitro toxicity, and intracellular detection. *Nano Lett* 5: 2123–2130.
- Levard C, Hotze EM, Colman BP, Dale AL, Truong L, Yang XY, Bone AJ, Brown GE, Tanguay RL, Di Giulio RT, Bernhardt ES, Meyer JN, Wiesner MR, Lowry GV. 2013. Sulfidation of silver nanoparticles: Natural antidote to their toxicity. *Environ Sci Technol* 47:13440–13448.
- Levard C, Hotze EM, Lowry GV, Brown GE. 2012. Environmental transformations of silver nanoparticles: Impact on stability and toxicity. *Environ Sci Technol* 46:6900–6914.
- Li XM, Schirmer K, Bernard L, Sigg L, Pillai S, Behra R. 2015. Silver nanoparticle toxicity and association with the alga *Euglena gracilis*. *Environ Sci Nano* 2:594–602.
- Linse S, Cabaleiro-Lago C, Xue WF, Lynch I, Lindman S, Thulin E, Radford SE, Dawson KA. 2007. Nucleation of protein fibrillation by nanoparticles. *Proc Natl Acad Sci U S A* 104:8691–8696.
- Liu HH, Cohen Y. 2014. Multimedia environmental distribution of engineered nanomaterials. *Environ Sci Technol* 48:3281–3292.
- Liu J, Pennell KG, Hurt RH. 2011. Kinetics and mechanisms of nanosilver oxysulfidation. *Environ Sci Technol* 45:7345–7353.
- Liu J, Legros S, Ma G, Veinot JGC, von der Kammer F, Hofmann T. 2012. Influence of surface functionalization and particle size on the aggregation kinetics of engineered nanoparticles. *Chemosphere* 97:918–924.
- Liu L, Sun M, Li Q, Zhang H, Yu K, Li M, Zhang C, Cao G, Yuang Y, Zhai H, Chen W, Alvarez PJJ. 2016. High-facet-energy CdS nanorods are more reactive but less cytotoxic than lower-facet-energy homologues of similar morphology. *Nano Lett* 16:688–694.
- Liu R, Zhang HY, Ji ZX, Rallo R, Xia T, Chang CH, Nel A, Cohen Y. 2013. Development of structure–activity relationship for metal oxide nanoparticles. *Nanoscale* 5:5644–5653.
- Lombi E, Donner E, Taheri S, Tavakkoli E, Jamting AK, McClure S, Naidu R, Miller BM, Scheckel KG, Vasilev K. 2013. Transformation of four silver/silver chloride nanoparticles during anaerobic treatment of wastewater and post-processing of sewage sludge. *Environ Pollut* 176:193–197.
- Lombi E, Donner E, Tavakkoli E, Turney TW, Naidu R, Miller BM, Scheckel KG. 2012. Fate of zinc oxide nanoparticles during anaerobic digestion of wastewater and post-treatment processing of sewage sludge. *Environ Sci Technol* 46:9089–9096.
- López-Serrano OA, Croteau MN, Stoiber TL, Tejamaya M, Römer I, Lead JR, Luoma SN. 2014. Does water chemistry affect the dietary uptake and toxicity of silver nanoparticles by the freshwater snail *Lymnaea stagnalis*? *Environ Pollut* 189:87–91.
- Lowry GV, Gregory KB, Apte SC, Lead JR. 2012a. Transformations of nanomaterials in the environment. *Environ Sci Technol* 46:6893–6899.
- Lowry GV, Espinasse BP, Badireddy AR, Richardson CJ, Reinsch BC, Bryant LD, Bone AJ, Deonarine A, Chae S, Therezien M, Colman BP, Hsu-Kim H, Bernhardt ES, Matson CW, Wiesner MR. 2012b. Long-term transformation and fate of manufactured Ag nanoparticles in a simulated large scale freshwater emergent wetland. *Environ Sci Technol* 46:7027–7036.
- Luoma SN, Khan FR, Croteau M-N. 2014. Bioavailability and bioaccumulation of metal-based engineered nanomaterials in aquatic environments: Concepts and processes. In Lead JR, Valsami-Jones E, eds, *Nanoscience and the Environment*, Vol 7—Frontiers of Nanoscience. Elsevier, Amsterdam, The Netherlands, pp 157–193.
- Luoma SN, Stoiber T, Croteau M-N, Römer I, Merrifield R, Lead JR. 2016. Effect of cysteine and humic acids on bioavailability of Ag from Ag nanoparticles to a freshwater snail. *NanoImpact* 2:61–69.
- Ma X, Wu Y, Jin S, Tian Y, Zhang X, Zhao Y, Yu L, Liang X-J. 2011. Gold nanoparticles induce autophagosome accumulation through size-dependent nanoparticle uptake and lysosome impairment. *ACS Nano* 5: 8629–8639.
- Ma H, Brennan A, Diamond SA. 2012. Photocatalytic reactive oxygen species production and phototoxicity of titanium dioxide nanoparticles are dependent on the solar ultraviolet radiation spectrum. *Environ Toxicol Chem* 31:2099–2107.

- Ma R, Levard C, Judy JD, Unrine JM, Durenkamp M, Martin B, Jefferson B, Lowry GV. 2013. Fate of zinc oxide and silver nanoparticles in a pilot wastewater treatment plant and in processed biosolids. *Environ Sci Technol* 48:104–112.
- Ma R, Stegemeier J, Levard C, Dale JG, Noack CW, Yang T, Brown GE, Lowry GV. 2014. Sulfidation of copper oxide nanoparticles and properties of resulting copper sulfide. *Environ Sci Nano* 1:347–357.
- Mahapatra I, Sun TY, Clark JRA, Dobson PJ, Hungerbuehler K, Owen R, Nowack B, Lead J. 2015. Probabilistic modelling of prospective environmental concentrations of gold nanoparticles from medical applications as a basis for risk assessment. *J Nanobiotechnol* 13:93.
- Mahler GJ, Esch MB, Tako E, Southard TL, Archer SD, Glahn RP, Shuler ML. 2012. Oral exposure to polystyrene nanoparticles affects iron absorption. *Nat Nanotechnol* 7:264–271.
- Majumdar S, Trujillo-Reyes J, Hernandez-Viezcas JA, White JC, Peralta-Videa JR, Gardea-Torresdey JL. 2016. Cerium biomagnification in a terrestrial food chain: Influence of particle size and growth stage. *Environ Sci Technol* 50:6782–9792.
- Manciualea A, Baker A, Lead JR. 2009. A fluorescence quenching study of the interaction of Suwannee River fulvic acid with iron oxide nanoparticles. *Chemosphere* 76:1023–1027.
- Marano F, Hussain S, Rodrigues-Lima F, Baeza-Squiban A, Boland S. 2011. Nanoparticles: Molecular targets and cell signalling. *Arch Toxicol* 85:733–741.
- McBeath TM, McLaughlin MJ. 2014. Efficacy of zinc oxides as fertilisers. *Plant Soil* 374:843–855.
- McNeil PL, Boyle D, Henry TB, Handy RD, Sloman KA. 2014. Effects of metal nanoparticles on the lateral line system and behaviour in early life stages of zebrafish (*Danio rerio*). *Aquat Toxicol* 152:318–323.
- Meermann B. 2015. Field-flow fractionation coupled to ICP-MS: Separation at the nanoscale, previous and recent application trends. *Anal Bioanal Chem* 407:2665–2674.
- Meisterjahn B, Wagner S, von der Kammer F, Hennecke D, Hofmann T. 2016. Silver and gold nanoparticle separation using asymmetrical flow-field flow fractionation: Influence of run conditions and of particle and membrane charges. *J Chromatogr A* 1440:150–159.
- Merrifield DL, Shaw BJ, Harper GM, Saoud IP, Davies SJ, Handy RD, Henry TB. 2013. Ingestion of metal-nanoparticle contaminated food disrupts endogenous microbiota in zebrafish (*Danio rerio*). *Environ Pollut* 174:157–163.
- Merrifield RC, Lead J. 2016. Preparation and characterization of three-layer, isotopically labelled core-shell nanoparticles: A tool for understanding mechanisms of bioavailability. *NanoImpact* 2:54–60.
- Merrifield RC, Wang ZW, Palmer RE, Lead JR. 2013. Synthesis and characterization of polyvinylpyrrolidone coated cerium oxide nanoparticles. *Environ Sci Technol* 47:12426–12433.
- Merrifield RC, Arkill KP, Palmer RE, Lead JR. 2017a. A high resolution study of dynamic changes of Ce₂O₃ and CeO₂ nanoparticles in complex environmental media. *Environ Sci Technol* 51:8014–8016.
- Merrifield RC, Stephan C, Lead J. 2017b. Determining the concentration dependent transformations of Ag nanoparticles in complex media: Using SP-ICP-MS and Au@Ag core shell nanoparticles as tracers. *Environ Sci Technol* 51:3206–3213.
- Merrifield RC, Stephan C, Lead JR. 2017c. Single-particle inductively coupled plasma mass spectroscopy analysis of size and number concentration in mixtures of monometallic and bimetallic (core-shell) nanoparticles. *Talanta* 162:130–134.
- Merrifield RC, Stephan C, Lead JR. 2018. Quantification of Au nanoparticle biouptake and freshwater algae using single cell-ICP-MS. *Environ Sci Technol* 52:2271–2277.
- Miao Q, Lyu Y, Ding D, Pu K. 2016. Semiconducting oligomer nanoparticles as an activatable photoacoustic probe with amplified brightness for in vivo imaging of pH. *Adv Mater* 28:3662–3668.
- Miller RJ, Bennett S, Keller AA, Pease S, Lenihan HS. 2012. TiO₂ nanoparticles are phototoxic to marine phytoplankton. *PLoS One* 7:7.
- Millaku A, Drobne D, Torkar M, Novak S, Remškar M, Pipan-Tkalec Ž. 2013. Use of scanning electron microscopy to monitor nanofibre/cell interaction in digestive epithelial cells. *J Hazard Mater* 260:47–52.
- Minghetti M, Drieschner C, Bramaz N, Schug H, Schirmer K. 2017. A fish intestinal epithelial barrier model established from the rainbow trout (*Oncorhynchus mykiss*) cell line, RTgutGC. *Cell Biol Toxicol* 33:539–555.
- Minghetti M, Schirmer K. 2016. Effect of media composition on bioavailability and toxicity of silver and silver nanoparticles in fish intestinal cells (RTgutGC). *Nanotoxicology* 10:1526–1534.
- Miralles P, Church TL, Harris AT. 2012. Toxicity, uptake, and translocation of engineered nanomaterials in vascular plants. *Environ Sci Technol* 46:9224–9239.
- Misra SK, Dybowska A, Berhanu D, Croteau MN, Luoma SN, Boccaccini AR, Valsami-Jones E. 2012. Isotopically modified nanoparticles for enhanced detection in bioaccumulation studies. *Environ Sci Technol* 46:1216–1222.
- Moore JD, Stegemeier JP, Bibby K, Marinakos SM, Lowry GV, Gregory KB. 2016. Impacts of pristine and transformed Ag and Cu engineered nanomaterials on surficial sediment microbial communities appear short-lived. *Environ Sci Technol* 50:2641–2651.
- Mouchet F, Landois P, Sarremejean E, Bernard G, Puech P, Pinelli E, Flahaut E, Gauthier L. 2008. Characterisation and in vivo ecotoxicity evaluation of double-wall carbon nanotubes in larvae of the amphibian *Xenopus laevis*. *Aquat Toxicol* 87:127–137.
- Mu Y, Wu F, Zhao Q, Ji R, Qie Y, Zhou Y, Hu Y, Pang C, Hristozov D, Giesy JP. 2016. Predicting toxic potencies of metal oxide nanoparticles by means of nano-QSARs. *Nanotoxicology* 10:1207–1214.
- Mudunkotuwa IA, Grassian VH. 2015. Biological and environmental media control oxide nanoparticle surface composition: The roles of biological components (proteins and amino acids), inorganic oxyanions and humic acid. *Environ Sci Nano* 2:429–439.
- Mueller NC, Nowack B. 2008. Exposure modeling of engineered nanoparticles in the environment. *Environ Sci Technol* 42:4447–4453.
- Mühling M, Bradford A, Readman JW, Somerfield PJ, Handy RD. 2009. An investigation into the effects of silver nanoparticles on antibiotic resistance of naturally occurring bacteria in an estuarine sediment. *Mar Environ Res* 68:278–283.
- Nabiev I, Mitchell S, Davies A, Williams Y, Kelleher D, Moore R, Gun'ko YK, Byrne S, Rakovich YP, Donegan JF, Sukhanova A, Conroy J, Cottell D, Gaponik N, Rogach A, Volkov Y. 2007. Nonfunctionalized nanocrystals can exploit a cell's active transport machinery delivering them to specific nuclear and cytoplasmic compartments. *Nano Lett* 7:3452–3461.
- Nations S, Wages M, Cañas JE, Maul J, Theodorakis C, Cobb GP. 2011. Acute effects of Fe₂O₃, TiO₂, ZnO and CuO nanomaterials on *Xenopus laevis*. *Chemosphere* 83:1053–1061.
- Navarro E, Baun A, Behra R, Hartmann NB, Filser J, Miao A-J, Quigg A, Santschi PH, Sigg L. 2008. Environmental behavior and ecotoxicity of engineered nanoparticles to algae, plants, and fungi. *Ecotoxicology* 17:372–386.
- Navarro DA, Kookana RS, Kirby JK, Martin SM, Shareef A, Du J, McLaughlin MJ. 2013. Behaviour of fullerenes (C-60) in the terrestrial environment: Potential release from biosolids-amended soils. *J Hazard Mater* 262:496–503.
- Navarro DA, Kookana RS, McLaughlin MJ, Kirby JK. 2016. Fullerol as a potential pathway for mineralization of Fullerene nanoparticles in biosolid-amended soils. *Environ Sci Technol Lett* 3:7–12.
- Nel A, Xia T, Madler L, Li N. 2006. Toxic potential of materials at the nanolevel. *Science* 311:622–627.
- Nel A, Mädler L, Velegol D, Xia T, Hoek EMV, Somadundaran P, Klaessig F, Castranova V, Thompson M. 2009. Understanding biophysicochemical interactions at the nano-bio interface. *Nat Mater* 8:543–557.
- Nowack B, Baalousha M, Bornhöft N, Chaudhry Q, Cornelis G, Cotterill J, Gondikas A, Hassellöv M, Lead J, Mitrano DM. 2015. Progress towards the validation of modeled environmental concentrations of engineered nanomaterials by analytical measurements. *Environ Sci Nano* 2:421–428.
- Nyberg L, Turco RF, Nies L. 2008. Assessing the impact of nanomaterials on anaerobic microbial communities. *Environ Sci Technol* 42:1938–1943.
- Odzak N, Kistler D, Behra R, Sigg L. 2015. Dissolution of metal and metal oxide nanoparticles under natural freshwater conditions. *Environ Chem* 12:138–148.
- Odzak N, Kistler D, Sigg L. 2017. Influence of daylight on the fate of silver and zinc oxide nanoparticles in natural aquatic environments. *Environ Pollut* 226:1–11.
- Omar FM, Aziz HA, Stoll S. 2014. Aggregation and disaggregation of ZnO nanoparticles: Influence of pH and adsorption of Suwannee River humic acid. *Sci Total Environ* 468–469:195–201.

- Organisation for Economic Co-operation and Development. 2007. Testing programme of manufactured nanomaterials. Paris, France. [cited 2018 May 28]. Available from: <http://www.oecd.org/chemicalsafety/nanosafety/testing-programme-manufactured-nanomaterials.htm>
- Pace HE, Rogers NJ, Jarolimek C, Coleman VA, Higgins CP, Ranville JF. 2011. Determining transport efficiency for the purpose of counting and sizing nanoparticles via single particle inductively coupled plasma mass spectrometry. *Anal Chem* 83:9361–9369.
- Pace HE, Rogers NJ, Jarolimek C, Coleman VA, Gray EP, Higgins CP, Ranville JF. 2012. Single particle inductively coupled plasma-mass spectrometry: A performance evaluation and method comparison in the determination of nanoparticle size. *Environ Sci Technol* 46:12272–12280.
- Paine PL, Moore LC, Horowitz SB. 1975. Nuclear envelope permeability. *Nature* 254:109–114.
- Pakarinen K, Akkanen J, Leppänen MT, Kukkonen JV. 2014. Distribution of fullerenes (nC60) between sediment and water in freshwaters. *Chemosphere* 108:320–325.
- Pan J-F, Buffet P-E, Poirier L, Amiard-Triquet C, Gilliland D, Joubert Y, Pilet P, Guibbolini M, de Faverney CR, Roméo M. 2012. Size dependent bioaccumulation and ecotoxicity of gold nanoparticles in an endobenthic invertebrate: The Tellinid clam *Scrobicularia plana*. *Environ Pollut* 168:37–43.
- Pang C, Selck H, Banta GT, Misra SK, Berhanu D, Dybowska A, Valsami-Jones E, Forbes VE. 2013. Bioaccumulation, toxicokinetics, and effects of copper from sediment spiked with aqueous Cu, nano-CuO, or micro-CuO in the deposit-feeding snail, *Potamopyrgus antipodarum*. *Environ Toxicol Chem* 32:1561–1573.
- Panté N, Kann M. 2002. Nuclear pore complex is able to transport macromolecules with diameters of about 39 nm. *Mol Biol Cell* 13:425–434
- Park KH, Chhowalla M, Iqbal Z, Sesti F. 2003. Single-walled carbon nanotubes are a new class of ion channel blockers. *J Biol Chem* 278:50212–50216.
- Petersen EJ, Huang QG, Weber WJ. 2008a. Ecological uptake and depuration of carbon nanotubes by *Lumbricus variegatus*. *Environ Health Perspect* 116:496–500.
- Petersen EJ, Huang Q, Weber WJ. 2008b. Bioaccumulation of radio-labeled carbon nanotubes by *Eisenia foetida*. *Environ Sci Technol* 42:3090–3095.
- Petersen EJ, Pinto RA, Zhang L, Huang Q, Landrum PF, Weber WJ. 2011. Effects of polyethyleneimine-mediated functionalization of multi-walled carbon nanotubes on earthworm bioaccumulation and sorption by soils. *Environ Sci Technol* 45:3418–3425.
- Pettibone JM, Liu J. 2016. *In situ* methods for monitoring silver nanoparticle sulfidation in simulated waters. *Environ Sci Technol* 50:11145–11153.
- Pettitt ME, Lead JR. 2013. Minimum physicochemical characterisation requirements for nanomaterial regulation. *Environ Int* 52:41–50.
- Piccapietra F, Sigg L, Behra R. 2012. Colloidal stability of carbonate-coated silver nanoparticles in synthetic and natural freshwater. *Environ Sci Technol* 46:818–825.
- Pradas del Real AE, Castillo-Michel H, Kaegi R, Sinnet B, Magnin V, Findling N, Villanova J, Carriere M, Santaella C, Fernandez-Martinez A, Levard C, Sarret G. 2016. Fate of Ag-NMs in sewage sludge after application on agricultural soils. *Environ Sci Technol* 50:1759–1768.
- Praetorius A, Scheringer M, Hungerbühler K. 2012. Development of environmental fate models for engineered nanoparticles—A case study of TiO₂ nanoparticles in the Rhine River. *Environ Sci Technol* 46:6705–6713.
- Praetorius A, Labille J, Scheringer M, Thill A, Hungerbühler K, Bottero JY. 2014. Heteroaggregation of titanium dioxide nanoparticles with model natural colloids under environmentally relevant conditions. *Environ Sci Technol* 48:10690–10698.
- Prasad A, Lead JR, Baalousha M. 2015. An electron microscopy based method for the detection and quantification of nanomaterial number concentration in environmentally relevant media. *Sci Total Environ* 537:479–486.
- Priester JH, Ge Y, Mielke RE, Horst AM, Moritz SC, Espinosa K, Gelb J, Walker SL, Nisbet RM, An Y-J. 2012. Soybean susceptibility to manufactured nanomaterials with evidence for food quality and soil fertility interruption. *Proc Natl Acad Sci U S A* 109:E2451–E2456.
- Puzyn T, Rasulev B, Gajewicz A, Hu X, Dasari TP, Michalkova A, Hwang H-M, Toropov A, Leszczynska D, Leszczynski J. 2011. Using nano-QSAR to predict the cytotoxicity of metal oxide nanoparticles. *Nat Nanotechnol* 6:175–178.
- Quik JTK, Lynch I, Van Hoecke K, Miermans CJH, De Schampelaere KAC, Janssen CR, Dawson KA, Stuart MAC, Meent D. 2010. Effect of natural organic matter on cerium dioxide nanoparticles settling in model fresh water. *Chemosphere* 81:711–715.
- Quik JTK, Vonk JA, Hansen SF, Baun A, Meent D. 2011. How to assess exposure of aquatic organisms to manufactured nanoparticles? *Environ Int* 37:1068–1077.
- Quik JTK, Stuart MC, Wouterse M, Peijnenburg W, Hendriks AJ, Meent D. 2012. Natural colloids are the dominant factor in the sedimentation of nanoparticles. *Environ Toxicol Chem* 31:1019–1022.
- Quik JTK, Velzeboer I, Wouterse M, Koelmans AA, Meent D. 2014. Heteroaggregation and sedimentation rates for nanomaterials in natural waters. *Water Res* 48:269–279.
- Ramsden CS, Smith TJ, Shaw BJ, Handy RD. 2009. Dietary exposure to titanium dioxide nanoparticles in rainbow trout (*Oncorhynchus mykiss*): No effect on growth, but subtle biochemical disturbances in the brain. *Ecotoxicology* 18:939–951.
- Ramsden CS, Henry TB, Handy RD. 2013. Sub-lethal effects of titanium dioxide nanoparticles on the physiology and reproduction of zebrafish. *Aquat Toxicol* 126:404–413.
- Ramskov T, Croteau M-N, Forbes VE, Selck H. 2015. Biokinetics of different-shaped copper oxide nanoparticles in the freshwater gastropod, *Potamopyrgus antipodarum*. *Aquat Toxicol* 163:71–80.
- Reed RB, Higgins CP, Westerhoff P, Tadjiki S, Ranville JF. 2012. Overcoming challenges in analysis of polydisperse metal-containing nanoparticles by single particle inductively coupled plasma mass spectrometry. *J Anal At Spectrom* 27:1093–1100.
- Rice CM, Burton GA, Wenning RJ, Ward H, Fernandes T, Lead J, Roberts A, van Genderen E. 2016. In Memoriam: Stephen J. Klaine. *Environ Toxicol Chem* 35:1607–1608.
- Ringwood AH, Levi-Polyachenko N, Carroll DL. 2009. Fullerene exposures with oysters: Embryonic, adult, and cellular responses. *Environ Sci Technol* 43:7136–7141.
- Rogers NJ, Franklin NM, Apte SC, Batley GE, Angel BM, Lead JR, Baalousha M. 2010. Physico-chemical behaviour and algal toxicity of nanoparticulate CeO₂ in freshwater. *Environ Chem* 7:50–60.
- Römer I, Gavin AJ, White TA, Merrifield RC, Chipman JK, Viant MR, Lead JR. 2013. The critical importance of defined media conditions in *Daphnia magna* nanotoxicity studies. *Toxicol Lett* 223:103–108.
- Römer I, Wang ZW, Merrifield RC, Palmer RE, Lead J. 2016. High resolution STEM-EELS study of silver nanoparticles exposed to light and humic substances. *Environ Sci Technol* 50:2183–2190.
- Sagee O, Dror I, Berkowitz B. 2012. Transport of silver nanoparticles (AgNMs) in soil. *Chemosphere* 88:670–675.
- Saleh NB, Aich N, Plazas-Tuttle J, Lead JR, Lowry GV. 2015. Research strategy to determine when novel nanohybrids pose unique environmental risks. *Environ Sci Nano* 2:11–18.
- Sani-Kast N, Scheringer M, Slomberg D, Labille J, Praetorius A, Ollivier P, Hungerbühler K. 2015. Addressing the complexity of water chemistry in environmental fate modeling for engineered nanoparticles. *Sci Total Environ* 535:150–159.
- Sayes CM. 2014. The relationships among structure, activity, and toxicity of engineered nanoparticles. *KONA Powder Particle J* 31:10–21.
- Schirmer K. 2014. Mechanisms of nanotoxicity. In Lead JR, Valsami-Jones E, eds, *Nanoscience and the Environment*, Vol 7—Frontiers of Nanoscience. Elsevier Science, Amsterdam, The Netherlands, pp 195–221.
- Schug H, Isaacson C, Sigg L, Ammann A, Schirmer K. 2014. Effect of TiO₂ nanoparticles and UV radiation on extracellular enzyme activity of intact heterotrophic biofilms. *Environ Sci Technol* 48:11620–11628.
- Schultz AG, Boyle D, Chamot D, Ong KJ, Wilkinson KJ, McGeer JC, Sunahara G, Goss GG. 2014. Aquatic toxicity of manufactured nanomaterials: Challenges and recommendations for future toxicity testing. *Environ Chem* 11:207–226.
- Schultz CL, Wamuch A, Tsyusko OV, Unrine JM, Crossley A, Svendsen C, Spurgeon DJ. 2016. Multigenerational exposure to silver ions and silver nanoparticles reveals heightened sensitivity and epigenetic memory in *Caenorhabditis elegans*. *Proc R Soc B* 283:20152911.

- Selck H, Handy RD, Fernandes TF, Klaine SJ, Petersen EJ. 2016. Nanomaterials in the aquatic environment: A European Union–United States perspective on the status of ecotoxicity testing, research priorities, and challenges ahead. *Environ Toxicol Chem* 35:1055–1067.
- Semenzin E, Lanzellotto E, Hristozov D, Critto A, Zabeo A, Giubilato E, Marcomini A. 2015. Species sensitivity weighted distribution for ecological risk assessment of engineered nanomaterials: The n-TiO₂ case study. *Environ Toxicol Chem* 34:2644–2659.
- Shaw BJ, Al-Bairuty G, Handy RD. 2012. Effects of waterborne copper nanoparticles and copper sulphate on rainbow trout (*Oncorhynchus mykiss*): Physiology and accumulation. *Aquat Toxicol* 116:90–101.
- Shaw BJ, Liddle CC, Windeatt KM, Handy RD. 2016. A critical evaluation of the fish early-life stage toxicity test for engineered nanomaterials: Experimental modifications and recommendations. *Arch Toxicol* 90:2077–2107.
- Shemetov AA, Nabiev I, Sukhanova A. 2012. Molecular interaction of proteins and peptides with nanoparticles. *ACS Nano* 6:4585–4602.
- Shoultz-Wilson A, Reinsch BC, Tsyusko OV, Bertsch PM, Lowry GV, Unrine JM. 2011. Role of particle size and soil type in toxicity of silver nanoparticles to earthworms. *Soil Sci Soc Amer J* 75:365–377.
- Skalska J, Strużyńska L. 2015. Toxic effects of silver nanoparticles in mammals—Does a risk of neurotoxicity exist? *Folia Neuropathol* 53:281–300.
- Soenen SJ, Nuytten N, De Meyer SF, De Smedt SC, De Cuyper M. 2010. High intracellular iron oxide nanoparticle concentrations affect cellular cytoskeleton and focal adhesion kinase-mediated signaling. *Small* 6: 832–842.
- Smith BM, Pike DJ, Kelly MO, Nason JA. 2015. Quantification of heteroaggregation between citrate-stabilized gold nanoparticles and hematite colloids. *Environ Sci Technol* 49:12789–12797.
- Smith CJ, Shaw BJ, Handy RD. 2007. Toxicity of single walled carbon nanotubes to rainbow trout (*Oncorhynchus mykiss*): Respiratory toxicity, organ pathologies, and other physiological effects. *Aquat Toxicol* 82:94–109.
- Smolders E, Degryse F. 2002. Fate and effect of zinc from tire debris in soil. *Environ Sci Technol* 36:3706–3710.
- Sondi I, Salopek-Sondi B. 2004. Silver nanoparticles as antimicrobial agent: A case study on *E. coli* as a model for gram-negative bacteria. *J Colloid Interface Sci* 274:177–182.
- Sovová T, Boyle D, Sloman KA, Pérez CV, Handy RD. 2014. Impaired behavioural response to alarm substance in rainbow trout exposed to copper nanoparticles. *Aquat Toxicol* 152:195–204.
- Starnes DL, Unrine JM, Starnes CP, Collin BE, Oostveen EK, Ma R, Lowry GV, Bertsch PM, Tsyusko OV. 2015. Impact of sulfidation on the bioavailability and toxicity of silver nanoparticles to *Caenorhabditis elegans*. *Environ Pollut* 196:239–246.
- Starnes DL, Lichtenberg SS, Unrine JM, Starnes CP, Oostveen EK, Lowry GV, Bertsch PM, Tsyusko OV. 2016. Distinct transcriptomic responses of *Caenorhabditis elegans* to pristine and sulfidized silver nanoparticles. *Environ Pollut* 213:314–321.
- Stern ST, Adisheshaiah PP, Crist RM. 2012. Autophagy and lysosomal dysfunction as emerging mechanisms of nanomaterial toxicity. *Part Fibre Toxicol* 9:20.
- Stoiber T, Croteau M-N, Romer I, Tejamaya M, Lead JR, Luoma SN. 2015. Influence of hardness on the bioavailability of silver to a freshwater snail after waterborne exposure to silver nitrate and silver nanoparticles. *Nanotoxicology* 9:918–927.
- Stone V, Hankin S, Aitken R, Aschberger K, Baun A, Christensen F, Fernandes T, Hansen SF, Hartmann NIB, Hutchinson G. 2010. Engineered nanoparticles: Review of health and environmental safety (ENRHES). *Project final report. European Commission. Brussels, Belgium.*
- Sun TY, Gottschalk F, Hungerbühler K, Nowack B. 2014. Comprehensive probabilistic modelling of environmental emissions of engineered nanomaterials. *Environ Pollut* 185:69–76.
- Sun TY, Mitrano DM, Bornhöft NA, Scheringer M, Hungerbühler K, Nowack B. 2017. Envisioning nano release dynamics in a changing world: Using dynamic probabilistic modeling to assess future environmental emissions of engineered nanomaterials. *Environ Sci Technol* 51:2854–2863.
- Taylor NS, Merrifield R, Williams TD, Chipman JK, Lead JR, Viant MR. 2016. Molecular toxicity of cerium oxide nanoparticles to the freshwater alga *Chlamydomonas reinhardtii* is associated with supra-environmental exposure concentrations. *Nanotoxicology* 10:32–41.
- Tharaud M, Gondikas AP, Benedetti MF, von der Kammer F, Hofmann T, Cornelis G. 2017. TiO₂ nanomaterial detection in calcium rich matrices by splCPMS. A matter of resolution and treatment. *J Anal At Spectrom* 32:1400–1411.
- Thit A, Banta GT, Selck H. 2015. Bioaccumulation, subcellular distribution and toxicity of sediment-associated copper in the ragworm *Nereis diversicolor*: The relative importance of aqueous copper, copper oxide nanoparticles and microparticles. *Environ Pollut* 202:50–57.
- Tlili A, Cornut J, Behra R, Gil-Allué C, Gessner MO. 2016. Harmful effects of silver nanoparticles on a complex detrital model system. *Nanotoxicology* 10:728–735.
- Tlili A, Jabiol J, Behra R, Gil-Allué C, Gessner MO. 2017. Chronic exposure effects of silver nanoparticles on stream microbial decomposer communities and ecosystem functions. *Environ Sci Technol* 51: 2447–2455.
- Tolaymat TN, El Badawy AM, Genaidy A, Scheckel KG, Luxton TP, Suidan M. 2010. An evidence-based environmental perspective of manufactured silver nanoparticle in syntheses and applications: A systematic review and critical appraisal of peer-reviewed scientific papers. *Sci Total Environ* 408:999–1006.
- Toncelli C, Mylona K, Kalantzi I, Tsiola A, Pitta P, Tsapakis M, Pergantis SA. 2017. Silver nanoparticles in seawater: A dynamic mass balance at part per trillion silver concentrations. *Sci Total Environ* 601–602:15–21.
- Tong Z, Bischoff M, Nies L, Applegate B, Turco RF. 2007. Impact of fullerene (C60) on a soil microbial community. *Environ Sci Technol* 41: 2985–2991.
- Tsiola A, Pitta P, Callol AJ, Kagiorgi M, Kalantzi I, Mylona K, Santi L, Toncelli C, Pergantis S, Tsapakis M. 2017. The impact of silver nanoparticles on marine plankton dynamics: Dependence on coating, size and concentration. *Sci Total Environ* 601:1838–1848.
- Unrine JM, Tsyusko OV, Hunyadi SE, Judy JD, Bertsch PM. 2010. Effects of particle size on chemical speciation and bioavailability of copper to earthworms (*Eisenia fetida*) exposed to copper nanoparticles. *J Environ Qual* 39:1942–1953.
- Unrine JM, Colman BP, Bone AJ, Gondikas AP, Matson CW. 2012a. Biotic and abiotic interactions in aquatic microcosms determine fate and toxicity of Ag nanoparticles. Part 1. Aggregation and dissolution. *Environ Sci Technol* 46:6915–6924.
- Unrine JM, Shoultz-Wilson WA, Zhurbich O, Bertsch PM, Tsyusko O. 2012b. Trophic transfer of Au nanoparticles from soil along a simulated terrestrial food chain. *Environ Sci Technol* 46:9753–9760.
- Van Hoecke K, Quik JTK, Mankiewicz-Boczek J, De Schampelaere KAC, Elsaesser A, Van der Meer P, Barnes C, McKerr G, Howard CV, Van De Meent D, Ryzdzynski K, Dawson KA, Salvati A, Lesniak A, Lynch I, Silversmit G, De Samber B, Vincze L, Janssen CR. 2009. Fate and effects of CeO₂ nanoparticles in aquatic ecotoxicity tests. *Environ Sci Technol* 43: 4537–4546.
- Vance ME, Kuiken T, Vejerano EP, McGinnis SP, Hochella Jr. MF, Rejeski D, Hull MS. 2015. Nanotechnology in the real world: Redeveloping the nanomaterial consumer products inventory. *Beilstein J Nanotechnol* 6:1769–1780.
- Wang DJ, Jaisi DP, Yan J, Jin Y, Zhou DM. 2015. Transport and retention of polyvinylpyrrolidone-coated silver nanoparticles in natural soils. *Vadose Zone J* 14:7.
- Wang J, Koo Y, Alexander A, Yang Y, Westerhof S, Zhang Q, Schnoor JL, Colvin VL, Braam J, Alvarez PJJ. 2013. Phytostimulation of poplars and *Arabidopsis* exposed to silver nanoparticles and Ag⁺ at sublethal concentrations. *Environ Sci Technol* 47:5442–5449.
- Wang P, Menzies NW, Lombi E, McKenna BA, Johannessen B, Glover CJ, Kappen P, Kopittke PM. 2013. Fate of ZnO nanoparticles in soils and cowpea (*Vigna unguiculata*). *Environ Sci Technol* 47:13822–13830.
- Wang P, Menzies NW, Dennis PG, Guo J, Forstner C, Sekine R, Lombi E, Kappen P, Bertsch P, Kopittke PM. 2016. Silver nanoparticles entering soils via the wastewater-sludge-soil pathway pose low risk to plants but elevated Cl concentrations increase Ag bioavailability. *Environ Sci Technol* 50:8274–8281.
- Wang Q, Ebbs SD, Chen Y, Ma X. 2013. Trans-generational impact of cerium oxide nanoparticles on tomato plants. *Metallomics* 5:753–759.
- Wang R, Yang W, Song Y, Shen X, Wang J, Zhong X, Li S, Song Y. 2015. A general strategy for nanohybrids synthesis via coupled competitive reactions controlled in a hybrid process. *Sci Rep* 5:9189.

- Wang Y, Li Y, Kim H, Walker SL, Abriola LM, Pennell KD. 2010. Transport and retention of fullerene nanoparticles in natural soils. *J Environ Qual* 39: 1925–1933.
- Wang Z, Li N, Zhao J, White JC, Qu P, Xing B. 2012. CuO nanoparticle interaction with human epithelial cells: Cellular uptake, location, export, and genotoxicity. *Chem Res Toxicol* 25:1512–1521.
- Ward JE, Kach DJ. 2009. Marine aggregates facilitate ingestion of nanoparticles by suspension-feeding bivalves. *Mar Environ Res* 68: 137–142.
- Warne MSJ, Batley GE, van Dam RA, Chapman JC, Fox D, Hickey CW, Stauber JL. 2015. Revised method for deriving Australian and New Zealand water quality guideline values for toxicants. Updated January, 2017. Department of Science, Information Technology and Innovation, Brisbane, QLD, Australia. [cited 2017 August 20]. Available from: <https://publications.csiro.au/rpr/pub?pid=csiro:EP159161>
- Wasmuth C, Rudel H, During RA, Klawonn T. 2016. Assessing the suitability of the OECD 29 guidance document to investigate the transformation and dissolution of silver nanoparticles in aqueous media. *Chemosphere* 144:2018–2023.
- Wiench K, Wohlleben W, Hisgen V, Radke K, Salinas E, Zok S, Landsiedel R. 2009. 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* 76:1356–1365.
- Wohlleben W, Neubauer N. 2016. Quantitative rates of release from weathered nanocomposites are determined across 5 orders of magnitude by the matrix, modulated by the embedded nanomaterial. *NanoImpact* 1:39–45.
- Wu W, Jiang C, Roy VAL. 2015. Recent progress in magnetic iron oxide–semiconductor composite nanomaterials as promising photocatalysts. *Nanoscale* 7:38–58.
- Wu YL, Putcha N, Ng KW, Leong DT, Lim CT, Loo SCJ, Chen X. 2013. Biophysical responses upon the interaction of nanomaterials with cellular interfaces. *Acc Chem Res* 46:782–791.
- Xia T, Kovoichik M, Liang M, Mädlar L, Gilbert B, Shi H, Yeh JI, Zink JI, Nel AE. 2008. Comparison of the mechanism of toxicity of zinc oxide and cerium oxide nanoparticles based on dissolution and oxidative stress properties. *ACS Nano* 2:2121–2134.
- Xiao Y, Vijver MG, Peijnenburg WJGM. 2018. Impact of water chemistry on the behavior and fate of copper nanoparticles. *Environ Pollut* 234: 684–691.
- Xiu Z, Liu Y, Mathieu J, Wang J, Zhu D, Alvarez PJ. 2014. Elucidating the genetic basis for *Escherichia coli* defense against silver toxicity using mutant arrays. *Environ Toxicol Chem* 33:993–997.
- Xiu Z-M, Ma J, Alvarez PJJ. 2011. Differential effect of common ligands and molecular oxygen on antimicrobial activity of silver nanoparticles versus silver ions. *Environ Sci Technol* 45:9003–9008.
- Yacobi NR, DeMaio L, Xie JS, Hamm-Alvarez SF, Borok Z, Kim KJ, Crandall ED. 2008. Polystyrene nanoparticle trafficking across alveolar epithelium. *Nanomed Nanotechnol Biol Med* 4:139–145.
- Yang X, Wang Q, Qu X, Jiang W. 2017. Bound and unbound humic acids perform different roles in the aggregation and deposition of multi-walled carbon nanotubes. *Sci Total Environ* 586:738–745.
- Yang Y, Alvarez PJJ. 2015. Sublethal concentrations of silver nanoparticles stimulate biofilm development. *Environ Sci Technol Lett* 2:221–226.
- Yang Y, Long CL, Li HP, Wang Q, Yang ZG. 2016. Analysis of silver and gold nanoparticles in environmental water using single particle-inductively coupled plasma-mass spectrometry. *Sci Total Environ* 563–564: 996–1007.
- Yin L, Cheng Y, Espinasse B, Colman BP, Auffan M, Wiesner M, Rose J, Liu J, Bernhardt ES. 2011. More than the ions: The effects of silver nanoparticles on *Lolium multiflorum*. *Environ Sci Technol* 45:2360–2367.
- Yin Y, Shen M, Tan Z, Yu S, Liu J, Jiang G. 2015. Particle coating-dependent interaction of molecular weight fractionated natural organic matter: Impacts on the aggregation of silver nanoparticles. *Environ Sci Technol* 49:6581–6589.
- Yue Y, Behra R, Sigg L, Freire PF, Pillai S, Schirmer K. 2015. Toxicity of silver nanoparticles to a fish gill cell line: Role of medium composition. *Nanotoxicology* 9:54–63.
- Yue Y, Behra R, Sigg L, Suter MJF, Pillai S, Schirmer K. 2016. Silver nanoparticle-protein interactions in intact rainbow trout gill cells. *Environ Sci Nano* 3:1174–1185.
- Yue Y, Li X, Sigg L, Suter MJ, Pillai S, Behra R, Schirmer K. 2017. Interaction of silver nanoparticles with algae and fish cells: A side by side comparison. *J Nanobiotechnol* 15:16.
- Zeng Z, Zheng W, Zheng H. 2017. Visualization of colloidal nanocrystal formation and electrode–electrolyte interfaces in liquids using TEM. *Acc Chem Res* 50:1808.
- Zhang J, Wages M, Cox SB, Maul JD, Li Y, Barnes M, Hope-Weeks L, Cobb GP. 2012. Effect of titanium dioxide nanomaterials and ultraviolet light coexposure on African clawed frogs (*Xenopus laevis*). *Environ Toxicol Chem* 31:176–183.
- Zhang LL, Hou L, Wang LL, Kan AT, Chen W, Tomson MB. 2012. Transport of fullerene nanoparticles (nC60) in saturated sand and sandy soil: Controlling factors and modeling. *Environ Sci Technol* 46:7230–7238.
- Zhang S, Jiang Y, Chen C-S, Creeley D, Schwehr KA, Quigg A, Chin W-C, Santschi PH. 2013. Ameliorating effects of extracellular polymeric substances excreted by *Thalassiosira pseudonana* on algal toxicity of CdSe quantum dots. *Aquat Toxicol* 126:214–223.
- Zhao C-M, Wang W-X. 2010. Biokinetic uptake and efflux of silver nanoparticles in *Daphnia magna*. *Environ Sci Technol* 44:7699–7704.
- Zhao CM, Wang WX. 2011. Comparison of acute and chronic toxicity of silver nanoparticles and silver nitrate to *Daphnia magna*. *Environ Toxicol Chem* 30:885–892.
- Zhao L, Peralta-Videa JR, Hernandez-Viezas JA, Hong J, Gardea-Torresday JL. 2012. Transport and retention behavior of ZnO nanoparticles in two natural soils: Effect of surface coating and soil composition. *J Nano Res* 17:229–242.
- Zhao L, Sun Y, Hernandez-Viezas JA, Hong J, Majumar S, Niu G, Duarte-Gardea M, Peralta-Videa JR, Gardea-Torresday JL. 2015. Monitoring the environmental effects of CeO₂ and ZnO nanoparticles through the life cycle of corn plants an in situ μ -XRF mapping of nutrients in kernels. *Environ Sci Technol* 49:2921–2928.
- Zhu H, Han J, Xiao J, Jin Y. 2008. Uptake, translocation, and accumulation of manufactured iron oxide nanoparticles by pumpkin plants. *J Environ Monit* 10:713–717.
- Zhu X, Zhu L, Chen Y, Tian S. 2009. Acute toxicities of six manufactured nanomaterial suspensions to *Daphnia magna*. *J Nanopart Res* 11:67–75.
- Zook JM, Maccuspie RI, Locascio LE, Melissa D, Halter MD, Elliott JT. 2011. Stable nanoparticle aggregates/agglomerates of different sizes and the effect of their size on hemolytic cytotoxicity. *Nanotoxicology* 5: 517–530.