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Exposure assessments of nanoparticles in aquatic environments – considerations, review and recommendations

SWEDISH CHEMICALS AGENCY

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Preface

The Swedish Chemicals Agency (KemI) has been assigned by the Swedish Government to produce a national action plan for a toxic-free everyday environment: Action plan for a toxic free everyday environment 2011 – 2014 – protect the children better.

Efforts are now going on in several areas, both in Sweden, within the EU and internationally and often in cooperation with other authorities. Reducing chemical risks in the everyday environment is one step towards attaining the Swedish Parliament´s environment quality objective A Non-Toxic Environment, which is the objective that KemI is responsible for. Within the framework of the action plan, KemI compiles knowledge in KemI´s report and PM series elaborated by experienced colleagues, researchers or consultants. In this way, KemI presents new and essential knowledge in publications which can be downloaded free of charge from the website www.kemikalieinspektionen.se

The action plan emphasizes the need for action to reach a high level of protection from the potential health and environmental risks caused by nanomaterials. The main objective of the present report was to outline and discuss important considerations for the exposure assessment of nanoparticles in the aquatic environment as part of the risk assessment. This includes also to discuss environmental conditions of importance for exposure assessment of nanoparticles, to review published exposure models for nanoparticles in aquatic environment and to outline important considerations for validation of exposure models for nanoparticles.

The report was written by Rickard Arvidsson, Division of Environmental Systems Analysis at Chalmers University of Technology, and Julián Gallego Urrea, Department of Chemistry and Molecular Biology at Gothenburg University together with Sverker Molander, Chalmers, and Martin Hassellöv, Gothenburg University. Project leader and contact at KemI was Yvonne Andersson. Stefan Gabring at KemI also provided valuable comments to the report. Responsible for the project at KemI was Agneta Falk-Filipsson, Head of Unit, Risk Reduction and Support.

The views expressed in the report are the author's own and do not necessarily reflect the view of KemI.

Content

Summary

Synthetic nanoparticles are new forms of chemical substances. They can be found in several different forms, such as free particles, surface bound and dissolved in liquid. Nanoparticles can also exist as free, individual particles or agglomerate consisting of multiple particles. This report discusses the assessment of possible risks of nanoparticles. Chemical risk is usually considered to consist of two elements: (1) Exposure to the substance, and (2) the substance's toxicity. So far, the risk-related research on nanoparticles has had a strong focus on the particles ' toxic effects. In this report, we would instead focus on how exposure to nanoparticles can be calculated and assessed, with focus on nanoparticles in water. In the report, we provide an initial background and definitions of nanomaterials and nanoparticles, and describe briefly a standard method of risk assessment of chemicals in the environment. Then we go through important considerations that should be made in the exposure assessment of nanoparticles. First we discuss three considerations related to the emissions of nanoparticles, namely the lack of data for annual production of nanoparticles, the importance of applying a substance flow perspective, and lack of data for so-called emission factors for nanoparticles of various products and materials. Furthermore, we discuss considerations for modeling of nanoparticles behavior in water, mainly by listing a number of key processes with large influence. These are agglomeration, sedimentation, and dissolution. Related to that, we discuss how natural organic materials, coatings and aging of particles can affect these processes. We note here three particle properties that are important in order to describe nanoparticles dispersion in water, in a similar way that the octanol-water partition coefficient and half-life is important to describe the fate of organic chemicals in the environment. For nanoparticles these are the particle size (a) and the density (ρ). We also identify a number of more complex parameters affecting particle behavior in the environment, but not only because of the different particle characteristics, but also depending on characteristics related to the environment. These are the collision efficiency (α), point of zero charge (pHpzc), Hamaker constant (A) and a so-called form factor (β) that affect the sedimentation. In addition to the general difficulty to measure or calculate these parameters they also co-vary. Furthermore, we make a review of 11 currently available exposure models for nanoparticles in aquatic environment. We note that the studies differ regarding modeling method, which sources of emissions that are included, the nanoparticles taken into account, estimated concentrations in the environment, and whether the results are presented as mass or particle concentration. Only two studies trying to model the nanoparticle exposure based on particle properties in a manner similar to standard methods for chemical risk assessment. The other modeling studies are instead based on data on flows of specific nanomaterials, and not on generic algorithms. Next, we describe a number of challenges that occur when measuring nanoparticles in the environment. Finally, we provide the following recommendations to ensure good exposure assessment of nanoparticles in the future:

1. Information of flows and stocks of nanoparticles in society need to be collected.
2. Emission factors would need to be developed for each product that makes use of nanoparticles.
3. Emissions should be reported both as mass and particle number until it becomes clearer which one is most relevant.
4. More research is needed in order to determine which particle properties need to be known in order to calculate the concentration of nanoparticles in the environment.
5. At least the particle size and particle size distribution, as well as the specific particle density should be reported.

6. More research is required to improve the experimental measurements of nanoparticles to be able to validate exposure models.

Sammanfattning

Syntetiska nanopartiklar utgör en ny form av kemiska ämnen. De kan finnas i flera olika former, såsom fria partiklar, ytbundna partiklar och partiklar lösta i vätska. Nanopartiklar kan vidare existera som fria, enskilda partiklar, eller som agglomerat bestående av flera partiklar som slagit sig samman. Denna rapport behandlar bedömning av möjliga risker med nanopartiklar. Kemisk risk brukar anses bestå av två element: (1) Exponeringen för ämnet, och (2) ämnets toxiska effekter. Hittills har den risk-relaterade forskningen om nanopartiklar haft ett kraftigt fokus på partiklarnas toxiska effekter. I denna rapport fokuserar vi istället på hur exponering av nanopartiklar kan beräknas och bedömas, med fokus på nanopartiklar i vatten. I rapporten ger vi inledningsvis en bakgrund och definitioner till nanomaterial och nanopartiklar, och beskriver kort en standardmetod för riskbedömning av kemikalier i miljön. Därefter går vi igenom viktiga hänsynstaganden som bör göras vid exponeringsbedömning av nanopartiklar. Vi diskuterar först tre hänsynstaganden relaterade till emissioner av nanopartiklar, nämligen bristen på data för årlig produktion av nanopartiklar, vikten av att tillämpa ett substansflödesperspektiv, och brist på data för så kallade emissionsfaktorer för nanopartiklar från olika produkter och material. Vidare diskuterar vi hänsynstaganden vid modellering av nanopartiklars beteende i vatten, främst genom att lista ett antal viktiga processer med stor påverkan. Dessa är agglomeration, sedimentation, och upplösning. Relaterat till det diskuterar vi hur naturligt organiskt material, ytbeläggningar och partiklarnas åldrande kan påverka dessa processer. Vi noterar här tre partikelegenskaper som är viktiga för att kunna beskriva nanopartiklarnas spridning i vatten, på liknande sätt som oktanol-vatten fördelningskoefficienten och halveringstid är viktiga för att beskriva spridningen av organiska kemikalier i miljön. För nanopartiklar är dessa storlek (a) och densitet (ρ). Vi identifierar också ett antal mer komplexa parametrar som påverkar partiklarnas beteende i miljön, som inte endast beror av olika partikelegenskaper, utan också av egenskaper relaterade till miljön. Dessa är kollisionseffektiviteten (α), nolladdningpunkten (pH_{pzc}), Hamakerkonstanten (A) och en så kallad formfaktor (β) som påverkar sedimentationen. Förutom att dessa är svåra att mäta eller beräkna generellt så samvarierar de. Vidare gör vi en genomgång av 11 kända publicerade exponeringsstudier av nanopartiklar i vatten. Vi konstaterar där att studierna skiljer sig åt gällande modelleringsmetod, vilka källor till utsläpp som inkluderats, vilka nanopartiklar som beaktats, beräknade koncentrationer i miljön, och huruvida resultaten presenteras som mass- eller partikelkoncentration. Endast två studier försöker modellera nanopartiklars exponering utifrån partikelegenskaper på ett sätt som påminner om standardmetoder för kemisk riskbedömning. De övriga studiernas modellering baseras istället på data på flöden av specifika nanomaterial, och inte på generella algoritmer. Därefter beskriver vi ett antal utmaningar som förekommer vid mätning av nanopartiklar i miljön. Avslutningsvis ger vi följande rekommendationer för att möjliggöra goda exponeringsbedömningar av nanopartiklar i framtiden:

1. Information om nanopartiklars flöden och lager i samhället behöver samlas in.
2. Emissionsfaktorer behöver tas fram för varje nanopartikel och produkt som används.
3. Rapportera resultat både som mass- och partikelkoncentration tills det utretts vilken som är mest relevant.
4. Mer forskning krävs för att avgöra vilka partikelegenskaper som behövs för att kunna beräkna en koncentration av nanopartiklar i miljön.
5. Åtminstone partikelstorlek och -fördelning samt partikeldensitet bör rapporteras.
6. Mer forskning krävs för att förbättra experimentella mätmetoder för nanopartiklar för att kunna validera exponeringsmodeller.

List of abbreviations

CNT – carbon nanotubes

DLS – dynamic light scattering

DLVO – Derjaguin, Landau, Verwey and Overbeek

EUSES – European Union System for the Evaluation of Substances

GIS – geographical information systems

LF2000-WQX – Low Flows 2000 Water Quality extension

NOM – natural organic matter

NTA – nanoparticle tracking analysis

PEC – predicted environmental concentration

PFA – particle flow analysis

PNEC – predicted no-effect concentration

RCR – risk characterization ratio

REACH – Registration, Evaluation, Authorization and restriction of Chemicals

SFA – substance flow analysis

1 Introduction

A number of authors have highlighted the potential environmental risks of nanomaterials, in particular nanoparticles (Colvin 2003; Royal Society 2004). It is the risks of nanoparticles that are in focus in this report. Scientists have suggested that nanoparticles may induce toxic effects due to their larger surface area-to-mass ratio and that new mechanisms for inducing toxic effects may be present for nanomaterials and nanoparticles (Nel et al. 2006). However, it is also clear that all nanoparticles are not hazardous, nor are all nanoparticles benign (Aitken et al. 2009). Since there are a large number of possible nanomaterials with widely differing physico-chemical, technical and biological properties, simple generalizations are not valid. Therefore, a number of studies have been conducted focusing on risks of individual nanomaterials or nanoparticles, for instance silver nanoparticles and nanomaterials (Wijnhoven et al. 2009), cerium and cerium oxide nanomaterials (Cassee et al. 2011), titanium dioxide nanoparticles (Johnson et al. 2011) and carbon nanotubes (Kohler et al. 2008).

The core of risk assessment of chemicals is to (1) estimate the exposure of a stressor to a certain endpoint of interest and (2) estimate a predicted no-effect concentration for that stressor with regard to the endpoint (Suter 1993; van Leeuwen and Vermeire 2007). The endpoint is often some measure of a response in an organism. This general approach has been suggested to be applicable for nanoparticles as well, albeit with some modifications (Wiesner et al. 2009). Predicted no-effect concentrations are derived from toxicological and ecotoxicological studies. So far, a considerable number of toxicological studies on nanoparticles have been conducted (Stone et al. 2009). There were, however, only about ten exposure assessment studies of nanoparticles until 2012, whereas thousands of studies on nanoparticle toxicity already existed in 2008 (Lubick 2008). Only some scientific literature has started to inquire the environmental fate and transformations of manufactured nanomaterials (Lowry et al. 2012; Nowack et al. 2012). It is thus clear that exposure assessment research is lagging behind in the area of risk assessment of nanoparticles. Assessing environmental exposure of nanoparticles thus remains a great scientific challenge and, consequently, there is a great need for models that can assess exposure pathways for nanoparticles in the environment. Modifications of exposure modelling of chemicals in order to become relevant for the case of nanoparticles have been suggested, including the development of new methods for assessing production (Arvidsson et al. 2011a; Hendren et al. 2011; Arvidsson et al. 2012). For example, Wiesner et al. (2009) and Sweet and Strohm (2006) have pointed out that emissions of nanoparticles in a prospective manner using substance life cycle approaches. In addition, new methods for fate modelling based on nanoparticle properties have been requested (Handy et al. 2008; Klaine et al. 2008).

1.1 Aim of the study

The aim of this study is to (1) *describe the environmental conditions of importance for exposure assessment of nanoparticles*, (2) *outline important considerations for exposure assessment of nanoparticles different from those of ordinary chemicals*, (3) *review the published exposure models for nanoparticles in aquatic environments and compare them,* and to (4) *suggest how to validate exposure models for nanoparticles.* Based on this, our final aim is to (5) *provide recommendations for exposure assessment of nanoparticles* for the future work of the Swedish Chemicals Agency and other regulatory bodies within this area, as well as suggestions for future research.

The first and second aims are dealt with in Chapter 2, the third aim is dealt with in Chapter 3, the fourth aim is dealt with in Chapter 4 and the fifth aim in Chapter 5. First, however, we provide the reader with a brief background to manufactured nanomaterials and nanoparticles and to their potential risks. For a more thorough introduction there are a number of books, articles and reports to which the reader may turn. One example is the report "Nanotechnology – big risks with small particles?" published by the Swedish Chemicals Agency (2007) which is available both in Swedish and in English. Another example is the report "Nanoscience and nanotechnologies: opportunities and uncertainties" by the Royal Society (2004).

The background to nanomaterials is followed by another brief background to how risk assessments of chemicals are conducted based on input data and modelling regarding a chemical substance. Exposure assessment of nanoparticles will then be explained in analogy to the case of chemicals in general.

1.2 Background to nanomaterials

Although he did not mention the term "nano" specifically, materials manufacture at the nano-scale were boosted in the now famous talk "There is Plenty of Room at the Bottom" by Richard P. Feynman in 1959, where he addressed the possibilities of manipulating single atoms as a more powerful form of synthetic chemistry. The term "nano", as in nanotechnology and nanomaterials, were first used by Norio Taniguchi who stated in 1974 that: "'Nano-technology' mainly consists of the processing of, separation, consolidation, and deformation of materials by one atom or one molecule." During the 2000's, there has been an increasing development of nanotechnology and products[1] containing nanomaterials and nanoparticles. For example, titanium dioxide nanoparticles are currently used, modified with a capping agent, in sunscreen to protect the skin from UV light (Barnard 2010) and is added to surfaces and bulk materials to provide self-cleaning ability (Chabas et al. 2008). Cerium dioxide nanoparticles are used as fuel additive in diesel but also in other applications (Park et al. 2008). Silver nanoparticles are used in various consumer products primarily for their antibacterial properties (Nowack et al. 2011). Iron nanoparticles have begun to be used for soil remediation (Zhang 2003). Carbon nanotubes have a number of applications such as electronics and composite materials, as do the nanomaterial graphene for which researchers were recently awarded the Nobel Prize in physics. The market is increasing for many nanomaterials such as several of the examples mentioned above (McWilliams 2008). A list of products which are claimed to contain nanomaterials by the producers can be found at the Woodrow Wilson's Project on Emerging Nanomaterials database (2009). The number of products included in this list rose to about 1300 during 2011.

A size range that at least one of the material's dimensions should stay within normally defines nanomaterials. This range may be 1-100 nm (British Standards Institution 2007; ISO 2008), 0.1-100 nm (Royal Society 2004), less than 100 nm (O'Brien and Cummins 2008), or less than 500 nm (Handy et al. 2008). It is interesting to note this diversity in size ranges. In addition, to be counted as a nanomaterial it is sometimes stated that the material must have properties different from those of the bulk form of the same chemical substance (Foss Hansen et al. 2007). A more detailed review of different definitions of the word nanomaterial, especially from a regulatory point of view, can be found in Lövestam et al. (2010). The

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[1] Note that the European legislation on Registration, Evaluation, Authorisation and restriction of Chemicals (REACH) differentiates between "articles" and "chemical products". An article is a product for which its physical properties are more important than its chemical ones. A chemical product is a chemical substance or mixture of substances. Here, however, we use the more common term "product" to denote all consumer products, including both the articles and chemical products of the REACH legislation.

European Commission published in October 2011 a recommended definition of nanomaterials: "'Nanomaterial' means a natural, incidental or manufactured material containing particles, in an unbound state or as an aggregate or as an agglomerate and where, for 50 % or more of the particles in the number size distribution, one or more external dimensions is in the size range 1 nm-100 nm" (European Commission 2011). It should be pointed out that the limit value of 50 % of the particle number size distribution as basis for the definition may introduce a number of problems that relate to the stability and transformation of the material and the association of the material to other materials in more complex products.

One way to categorize nanomaterials is through the framework developed by Foss Hansen et al. (2007), which is based on how the nanomaterials are physically structured and ordered. It includes three main categories, which are (1) bulk nanomaterials, (2) nanomaterials that constitute surfaces and (3) nanoparticles (Figure 1). The nanoparticles can then be divided into the categories airborne, surface-bound, suspended in a liquid and suspended in a solid. Jiang et al. (2009) presents another framework for categorization of nanoparticles, which include free particles, van der Waals bonded agglomerates and covalently bonded (sintered) aggregates. Another way to categorize nanomaterials (in particular nanoparticles) is on basis of their chemical composition. This was done by, for instance, Ju-Nam and Lead (2008), who included four different categories based on chemical composition: (1) carbon nanomaterials, such as fullerenes and carbon nanotubes, (2) metal oxide nanoparticles, such as titanium dioxide and zinc oxide, (3) metal nanoparticles, such as silver and iron nanoparticles, (4) others, such as quantum dots and nanopolymers.

In contrast to these definitions, the idea that defining nanomaterials is neither feasible nor wanted, and that current definitions lack scientific justification, has been put forth by Maynard (2011). The author further argues that instead of trying to establish a definition of nanomaterials suitable for all cases, individual risk-related properties of specific compounds should be considered. It can also be added that the very large number of combinations of shapes, core chemical compositions and different types of coating(s), which much influence properties of the nanoparticles, make categorisations challenging.
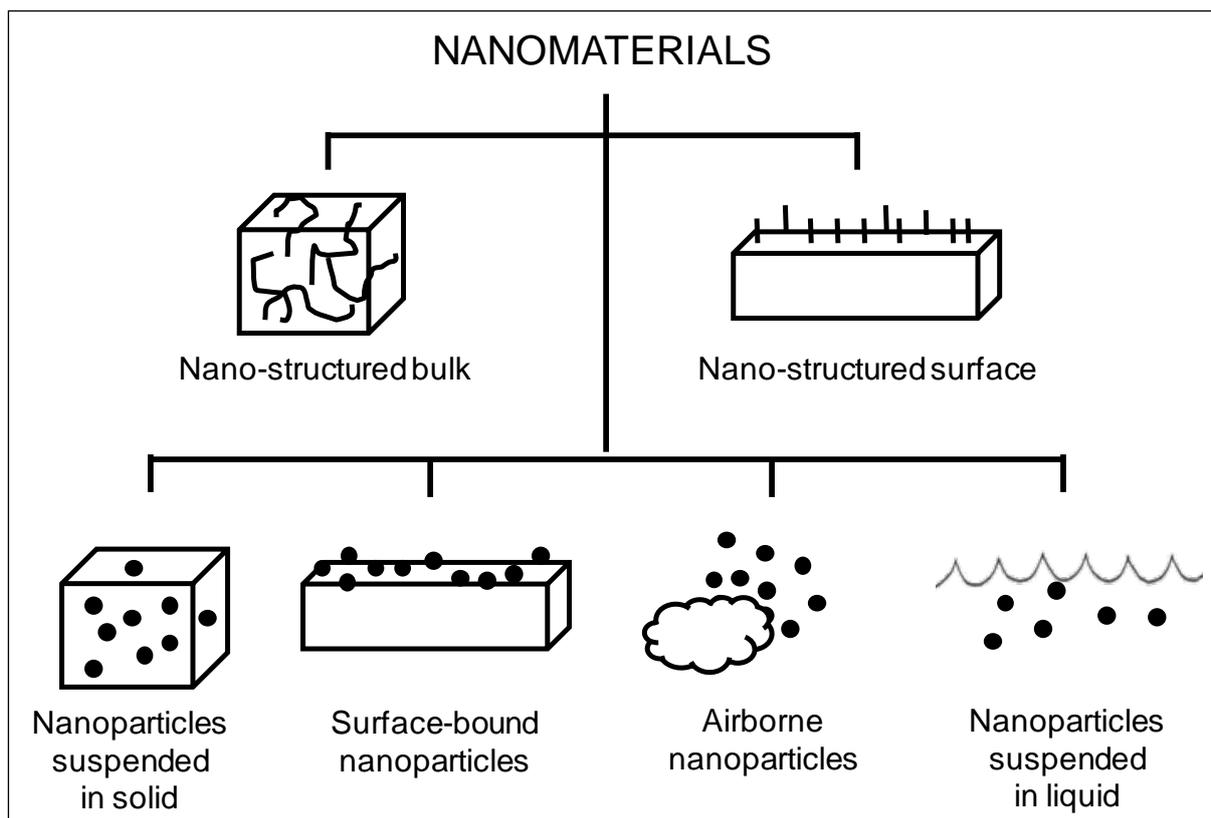
Figure 1. Categorization framework for nanomaterials modified from Foss Hansen et al. (2007).

1.3   Risk assessment of chemicals

The theoretical basis for this work is risk assessment of chemicals (van Leeuwen and Vermeire 2007; European Chemicals Agency 2011a). In this chapter, we give the reader a brief overview of this method in order to facilitate a better understanding of the rest of the report which focus only on the aquatic environment exposure assessment part. In order to conduct a risk assessment, the chemical under investigation must be identified. A risk assessment of chemicals is typically based on three different steps (Figure 2).

The first step in a risk assessment of a chemical substance is the exposure assessment. In this step, a quantitative estimation of how much of the stressor the endpoint is exposed to is conducted (European Chemicals Agency 2008a). For aquatic organisms, exposure to chemicals is often defined in terms of mass concentration in the surrounding water, although it is debated whether mass concentration is the most optimal exposure metric for the case of nanoparticles (Arvidsson et al. 2011b). The result of an exposure assessment is a predicted environmental concentration (PEC), which is often derived from mathematical models. Exposure is assessed by first assessing emissions and then model the environmental fate and uptake of the chemical. The emission assessment part aims at determining the emissions of the stressor from society to the environment and the fate modelling part aims at determining what happens to the stressor in the environment in terms of transport, partitioning, degradation and transformations.

The second step of a risk assessment is the hazard assessment step (sometimes referred to as effect assessment) (European Chemicals Agency 2011b). In this step, toxicological and ecotoxicological data are used in order to derive a predicted no-effect concentration (PNEC). The PNEC is the concentration of a chemical in any compartment below which unacceptable effects on the aquatic ecosystem and its organisms will most likely not occur during long term

or short term exposure. The PNEC is measured as mass concentration of the stressor in the surrounding environmental medium, although there are doubts that this is the most suitable effect metric for the case of nanoparticles (Dhawan et al. 2009). The third step is called risk characterization (European Chemicals Agency 2008b). In this step, the PEC and PNEC are compared in order to estimate the risk. Often, the PEC is divided by the PNEC, forming a so-called risk characterization ratio, here denoted RCR:

$$RCR = \frac{PEC}{PNEC} \begin{cases} \geq 1 : \text{risk not controlled} \\ < 1 : \text{risk controlled} \end{cases}$$

(1)

If the RCR is higher than or equal to one, it indicates that the PEC is higher than the PNEC, meaning that the risk is not controlled. If the RCR is lower than one, and that the PEC is lower than the PNEC, it indicates that the risk is controlled.
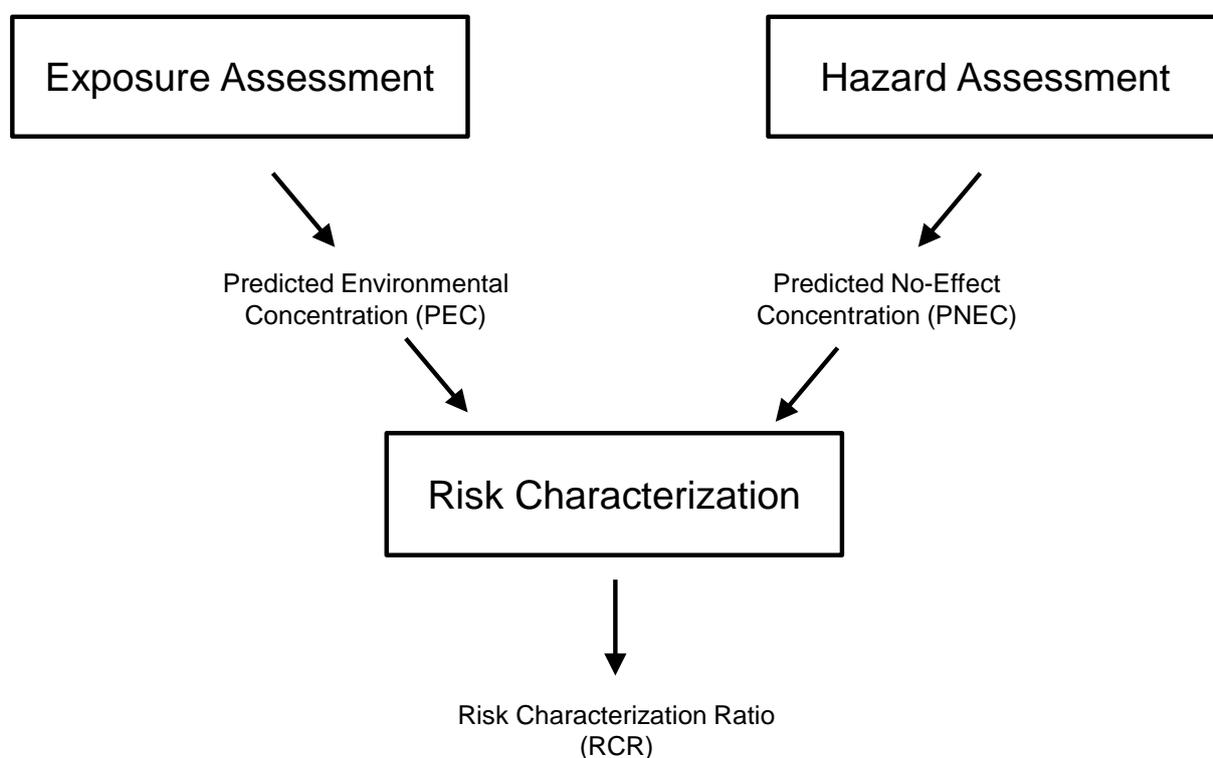
Figure 2. An illustration of the framework of risk assessment of chemicals. Modified from van Leeuwen and Vermeire (2007) and European Chemicals Agency (2011a).

Since this report is about how to conduct the exposure assessment step for the case of nanoparticles, the exposure assessment of chemicals is described in more detail here to provide a thorough background. Central in exposure models of chemicals are fate models, which are a valuable way of compiling knowledge and predicting the behaviour of a contaminant in various environmental compartments, based on a series of mathematical equations (European Chemicals Agency 2010). Implementing and exploring a fate model enables an assessment of the relative importance of different processes for the environmental fate of the chemical (Scheringer 2008).

MacLeod et al. (2010) describe the basic principles of common multimedia fate models for chemicals. The principal idea is to divide the environment into well-mixed, homogeneous

boxes, each one representing one compartment, such as different types of soil, sediment, water and air. In the basic models, each compartment is assumed to be in a steady state with its neighbouring compartments, the system is closed and in its simplest case ultimate losses due to biodegradation, abiotic degradation or sedimentation are ignored. Based on the fundamental principle of mass conservation, mass-balance equations are then employed for calculating the expected distribution of a chemical in the model system. More advanced, higher tier models, also allow the description of systems in non-equilibrium state that includes continuous or fluctuating inputs (emission) and loss functions (due to for instance volatilization and biodegradation). Furthermore, these models can be delimited in different ways, including different media and spatio-temporal scales. A validated fate model will provide results that help answering questions that are crucial for the exposure assessment of a chemical. Ultimately, they allow for an estimation of which environmental compartments that are expected to contain the highest concentrations and which concentrations that occur on different spatial scales. These models can also be implemented geographically referenced using geographical information systems (GIS) (Pistocchi et al. 2010). Validated fate models and exposure assessment approaches that incorporate them allow to predictively assess the consequences of additional emissions and of risk mitigation measures such as additional clean-up steps in wastewater treatment plants. Natural systems are often slow to respond. For example, the residence time of persistent pollutants such as mercury can be decades or longer in aquatic compartments. Therefore, it is almost impossible to explore and compare different risk mitigation measures experimentally in the real world.

In the context of the chemicals legislation REACH, predicted environmental concentrations of chemicals are usually estimated using simple steady-state multi-media models (European Chemicals Agency 2010). The following processes are considered during the fate assessment (Figure 3):

1. Adsorption to aerosol particles (gas-aerosol partitioning).
2. Partitioning between air and water (volatilisation).
3. Partitioning between solids and water in soil, sediment and suspended matter (adsorption and desorption).
4. Partitioning between water/solids and biota (bioconcentration).
5. Biomagnification within biota.
6. Dilutions, transport, deposition processes.
7. Biotic and abiotic transformation processes.

In order to describe these processes, the following minimum information is required for a standard chemical substance (an ordinary neutral non-hydrolysing organic chemical) according to the technical guidance on REACH (European Chemicals Agency 2010): molecular weight (M), water solubility, vapour pressure (p_{vap}), biodegradability (half-life, τ) and a number of partition coefficients (solids-water for sediment and sewage sludge, soil-water, suspended matter-water and sediment-water). From the EUSES model, partitioning coefficients can be obtained from some basic physico-chemical parameters, most importantly the octanol-water partition coefficient (K_{ow}). Using the K_{ow}, EUSES will generate partition coefficients for all the important exchange interfaces in the multi-media model, which drives the chemical fate and therefore the compartment concentrations. In other words, PECs of a chemical in different compartments are calculated as a function of a number of chemical properties. Of course, the emissions of the chemical (E) will also affect the PEC. Mathematically, PEC can then be described as a function of the emissions and a number of properties of the chemical:

$$\text{PEC} = f\left(E, M, p_{vap}, t, K_{ow, \llcorner}\right) \tag{2}$$

In the case of nanoparticles, the question is which properties that affect the PEC in a similar manner as the K_{ow} and the others do for non-nanoparticulate chemicals. It is unclear if the properties listed in Eq. 2 are relevant to describe the fate of nanoparticles as well, or if other properties are more important. This will be further discussed below.
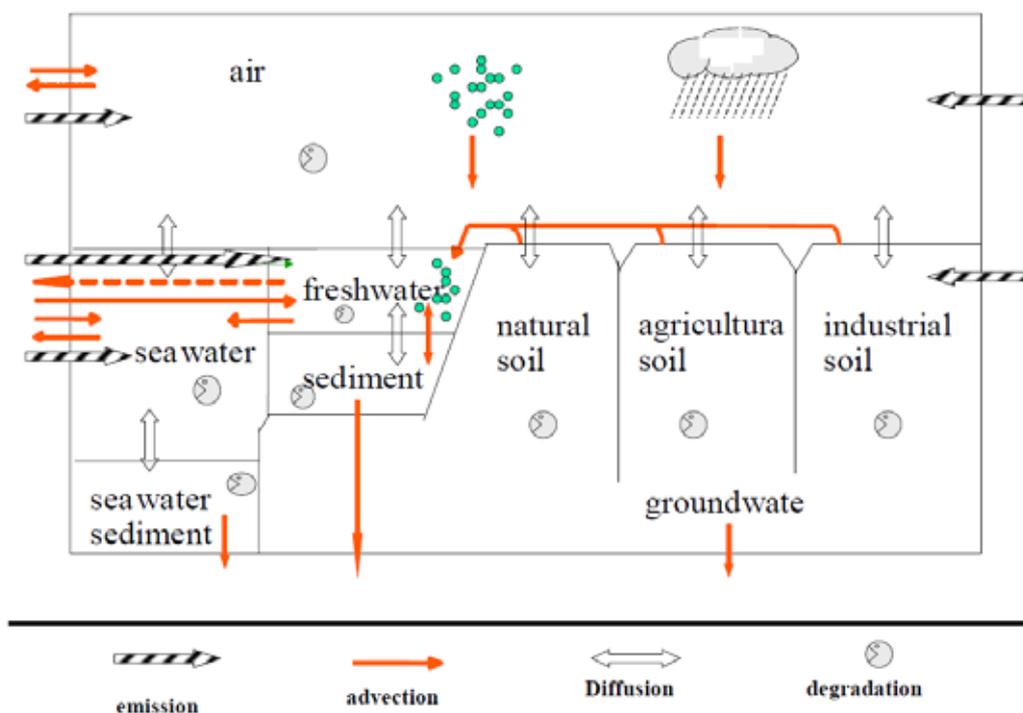

Figure 3. Environmental compartments and mass flows that are considered during the fate and exposure assessment according to EUSES. Obtained from European Chemicals Agency (2010).

2 Considerations for exposure modelling of nanoparticles in water

The following chapter outlines the critical issues that need consideration for adapting current exposure assessment approaches to engineered nanoparticles. First, issues to consider regarding emission assessment are discussed. Second, issues to consider regarding fate assessment of nanoparticles are discussed. Throughout this chapter, we will connect back to Section 1.3 and discuss how and to which extent the emission assessment, and in particular, the fate modelling of nanoparticles may require modifications compared to the risk assessment procedure for chemicals.

2.1 Considerations for emission assessment of nanoparticles

Assessing emissions to the environment is an important step in risk and exposure assessments of chemical substances, which takes place prior to fate modelling (van Leeuwen and Vermeire 2007). Even for quite well-known chemical substances the estimation of overall emissions poses serious difficulties. So although emissions of nanoparticles are in principle not different from those of chemicals not in nano form, the novelty of nanoparticles brings a number of challenges, on top of the general problem of estimation of emissions. Three issues for

consideration regarding the emission assessment of nanoparticles are highlighted here: the magnitude of nanoparticle emissions, the substance flow perspective and emission factors.

Since products containing nanoparticles are in most cases new on the market, one of the major challenges in assessing the exposure of engineered nanoparticles have been to estimate the magnitude of emission from society to the environment (Arvidsson et al. 2011a, 2012). Today, there is no monitoring of production rates, societal stocks and emissions of nanoparticles. The most well-known database for products containing nanoparticles is the Woodrow Wilson's Project on Emerging Nanotechnologies database. However, this database includes products based on the producer's claim that it contains nanomaterials, which may not always be correct. Products that do not include nanoparticles may be mistakenly included, and products that contain nanoparticles may not be included. In addition, no information regarding the exact content of nanoparticles or nanomaterials is provided. There exist only few sources of basic data regarding nanoparticle production, see for instance reports by the Royal Society (2004), UNEP (2007), Pérez et al. (2005), NICNAS (2010), BCC (2008) and the Silver Institute (2008). However, these basic sources do not always describe the methods applied and the sources used to obtain the data, and it is thus often difficult to assess the validity of the data. Considering this fundamental uncertainty, risk analysts have often had to turn to personal communication in order to obtain data on nanoparticle production rates (Blaser et al. 2008; Gottschalk et al. 2009). The results from these somewhat uncertain sources vary considerably, for instance 700 - 60 000 ton/years for the global emissions of titanium dioxide nanoparticles, 20 -10 000 ton/year for the global emissions of zinc oxide nanoparticles and 4 - 600 ton/year for the global emissions of silver nanoparticles (Gottschalk et al. 2009). Applying probabilistic methods as in Gottschalk et al. (2010), or scenarios as in Arvidsson et al. (2011a, 2012) may be a way to account for these variations, but can only partly compensate for the few and diverging sources of data. It is clear that more information regarding production of nanoparticles must be obtained in order to conduct accurate exposure assessments.

A number of authors have stated the importance of applying a substance life cycle perspective when estimating emissions of nanoparticles (Royal Society 2004; Sweet and Strohm 2006; Lubick 2008), as is also recommended for exposure assessment of chemicals under REACH (European Chemicals Agency 2008a). This means that emissions are investigated during the raw material acquisition, production, use and waste handling of the products in which the nanoparticles are constituents. Substance flow analysis has previously been successfully used to model emissions of hazardous chemicals (van der Voet 2002). Until now, however, few of the exposure assessments conducted on nanoparticles have included substance flow modelling in society. Instead, it has often been assumed that the total annual production of nanoparticles is emitted to the environment within that same year (Boxall et al. 2007; Mueller and Nowack 2008; Gottschalk et al. 2009). Using annual production as proxy for emissions is a reasonable approximation for products with dissipative use, such as sunscreen. However, for products that form stocks in society and have much longer life times, such as concrete and paint, the annual production may be very different from the annual emissions. It is thus important when modelling emissions to also include stocks of products in which nanoparticles are constituents, since it is from the surface of the societal stock of concrete and paint that emissions of nanoparticles from such products will occur. One problem here is the lack of knowledge regarding the size of material stocks in society, which exist only for a few well-studied materials, for instance in-use cement (Kapur et al. 2008).

Even with data on the stocks of products, estimating emissions also requires knowledge about emission factors. For products that form a stock in society, a certain amount of nanoparticles

may be emitted per unit of time. This amount will depend on (1) the concentration of nanoparticles in the stock, (2) the surface area of the stock, (3) if the surface is exposed to for instance wind, water or wear and (4) how strongly the particles are bound to the surface. In order to estimate emissions of nanoparticles from societal stocks, emission factors that include the aspects (1-4) mentioned above must be derived. Currently, only a small number of studies on nanoparticle emissions from products exist, and to our knowledge only five studies of titanium dioxide and silver nanoparticle emissions to the environment have been conducted. Kaegi et al. (2008) estimated the emissions of titanium dioxide nanoparticles from painted facades in the unit µg per litre of runoff from the facade. Hsu and Chein (2007) also estimated titanium dioxide nanoparticle emissions, but from wood, polymer and tile sprayed with nanoparticles. The output unit of the measurements was number of particles per unit of time and per sprayed surface area. Emissions of silver nanoparticles from antibacterial paint used on outdoor facades was measured by Kaegi et al. (2010), with both µg per square meter and µg per litre of runoff from the façade as output unit. Finally, three studies on emissions of silver nanoparticles from textiles have been conducted. Geranio et al. (2009) and Swedish Chemicals Agency (2011) studied emissions from different fabrics and Benn and Westerhoff (2008) emissions from socks, all measuring the amount of silver emitted per amount of textile as percentage of the total silver content. It is notable that such data exist only for a very limited number of nanoparticles and media. The results are also presented in different units, some of which are not easily compatible with fate models. More studies in this field are thus required in order to produce relevant emission data for exposure assessment purposes, since without data on emissions no estimation of PECs for nanoparticles can be made.

2.2 Considerations for fate modelling of nanoparticles

The specific behaviour of nanoparticles in suspension differs fundamentally from those of non-nanoparticulate chemicals. This gives rise to new challenges for the development and implementation of fate models. Nanoparticles in aqueous dispersions are aquatic colloids and hence, contrary to chemicals, are never in thermodynamic equilibrium (Elimelech et al. 1995). Several relevant fate processes for nanoparticles that are described below, such as agglomeration, sedimentation, particle deposition and dissolution are best described as rates of change rather than equilibrium. These kinetic aspects pose a challenge to conventional fate modelling, since the standard multi-media fate models are based on thermodynamic equilibrium concepts (Section 1.3). In the following, equations describing particle behaviour in water are presented. From these equations, fate-relevant nanoparticle properties determining their behaviour in water can be identified. In essence, nanoparticles in water will behave as colloids (Handy et al. 2008), and consequently much of the content of this chapter is based on material from the field of colloid chemistry. Figure 4 constitutes a graphical summary of the chapter and an illustration of the colloidal system of a water body exposed to manufactured nanoparticles.
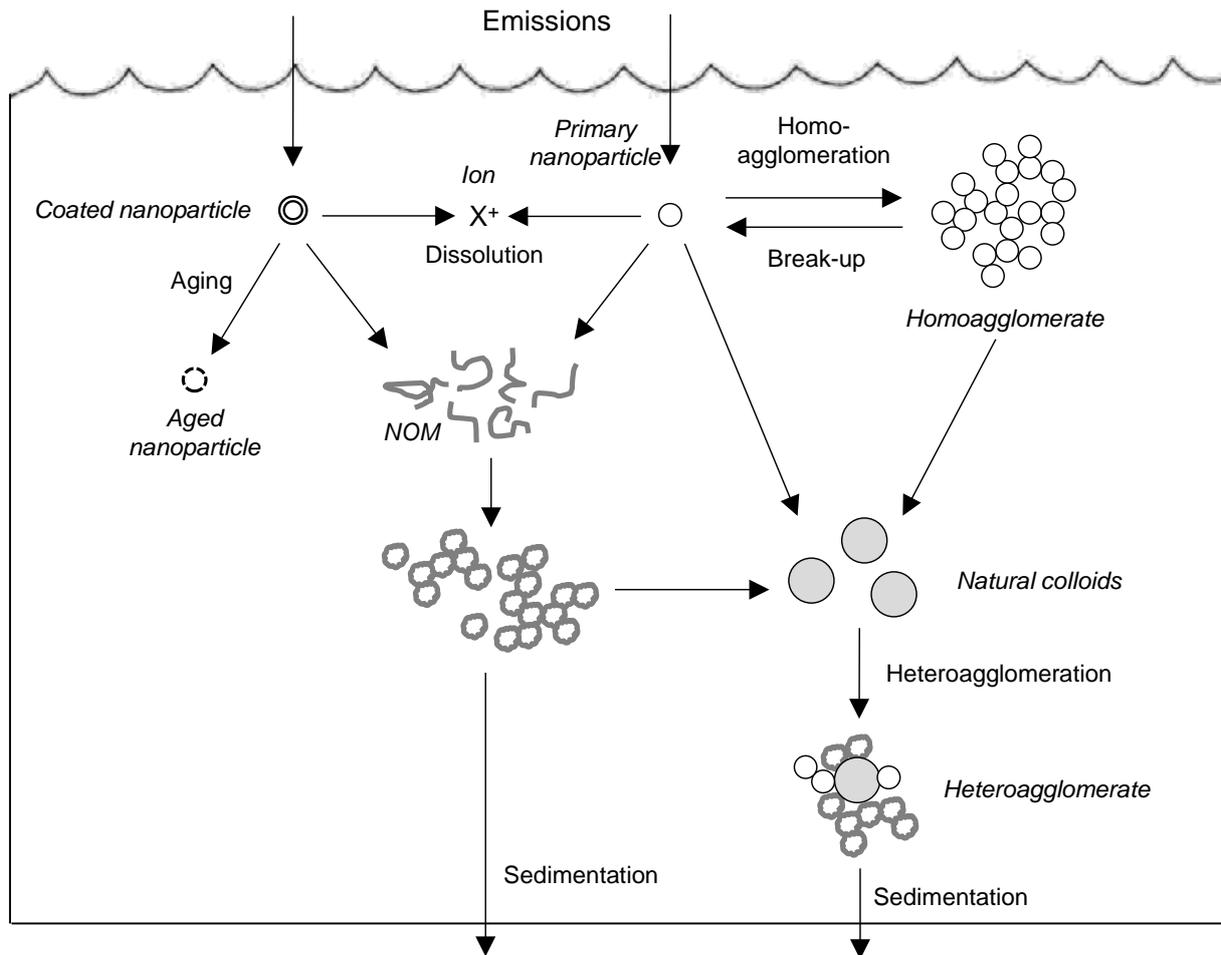
Figure 4. Schematic illustration of important fate processes for nanoparticles in the aquatic environment, including homoagglomeration, break-up, dissolution, sedimentation, reaction with natural organic matter (NOM) and natural colloids (the later also called heteroagglomeration), coating and aging. As can be seen, this system offers considerable complexity.

2.2.1 Agglomeration

Nanoparticles in waters will behave as colloids in the sense that they will be subject to diffusion with a diffusion coefficient related to their hydrodynamic size, and they will approach and collide with each other. Sometimes, these collisions will lead to attachment. Two particles will form one. There is a well-developed theory of kinetic agglomeration of particles by Smoluchovski (1917). According to this theory, agglomeration of particles can occur in three ways. The first is called perikinetic agglomeration, and describes the agglomeration of nanoparticles due to collisions arising from the particles movement due to diffusion. The second is called orthokinetic agglomeration, and describes the agglomeration of nanoparticles due to collisions arising from the movement of the water, i.e. the shear flow of the water. The third is differential settling, and describes the agglomeration of particles arising from the difference in particle movement due to sedimentation. According to Smoluchovski (1917), the agglomeration can be described by the following equation:

$$\frac{dn_j}{dt} = \frac{1}{2} \sum_{i=1}^{i=j-1} a_{i,j-i} K_{i,j-i} n_i n_{j-i} - n_j \sum_{i=1}^{i=\yen} a_{i,j} K_{i,j} n_i$$

(3)

where n_j is the particle number concentration of particle j (i.e. if j equals three, then particle j consists of three primary particles that have agglomerated), $\alpha_{i,j}$ and $\alpha_{i,j-i}$ are collision efficiencies, $K_{i,j}$ and $K_{i,j-i}$ are rate constants. The first term on the right side of the equation describes the formation of particle j through agglomeration of particles i and $j-i$. The second term describes the loss of particle j through agglomeration with other particles i. The rate constant K can be calculated according to the following equation:

$$K_{i,j} = \frac{2k_{B}T}{3m} \frac{(a_i + a_j)^2}{a_i a_j} + \frac{4}{3}G(a_i + a_j)^3 + \left(\frac{2\rho g}{9m}\right)(r_p - r_w)(a_i + a_j)^3(a_i - a_j)$$

(4)

where the first term is the perikinetic agglomeration, the second term is the orthokinetic agglomeration and the third term is differential settling, k_B is the Boltzmann constant, T is the temperature, μ is the fluid viscosity, a_i and a_j are the radii of particles i and j respectively, G is the shear rate, g is the gravitational acceleration and ρ_p and ρ_w are the densities of the particles and water, respectively. From Eq. 4, we can identify a number of nanoparticle properties that affect the PEC of nanoparticles. Note first that a PEC for nanoparticles can be derived from Eq. 3 by summarizing all particle sizes j and solving the differential equation. In analogy to Eq. 2 for the case of chemicals, we can then write the following equation for nanoparticle agglomeration:

$$PEC = f(a, a, r_p)$$

(5)

which states that the PEC of nanoparticles in water is a function of the collision efficiency (α), the particle size (a) and the particle density (ρ_p). Note also that in addition to these nanoparticle properties, a number of properties related to the water also affect the agglomeration behaviour. The particle size will affect the extent of which the particles diffuse, are carried by water and settle, and thus influence the agglomeration from these processes. The density of the particles affect the extent of settling and thus also the agglomeration due to differential settling. The particle size and density are basic properties that are possible to measure. The collision efficiency, however, is a more complex property that depends heavily on both particle properties but also properties of the water, as will be explained below.

2.2.2 Collision efficiency

The collision efficiency is a parameter that tells which fraction of the collisions that lead to attachment. That is, if the collision efficiency is equal to 1, all collisions will lead to attachment. If the collision efficiency is 0, no collisions will lead to attachment. The collision efficiency generally has a considerable impact on the agglomeration of particles.

Some particles in water will develop hydroxyl groups and some other protonated groups on the surface. These groups may be de-protonated depending on the pH of the medium. For example, at low pH the particles exhibit full protonation, which provides a positive charge, while at high pH the opposite holds true. At both extreme situations, all particles have the same strong charge and this leads to repulsion between particles. As a result, particles repel each other during close encounters and do not often fully collide close enough so that particles can be attracted by the short-range attractive van der Waals forces that are required for such an attachment. This mechanism of stabilization is called electrostatic stabilization, and leads to a low value of the collision efficiency. At pH values between both extremes there is a point where equal amounts of positive and negative charges are present and this pH value is called the point of zero charge (pH_{pzc}). Around the point of zero charge, rapid

agglomeration is often observed since there are no electrostatic repulsions preventing the particles from agglomerating. The value of the collision efficiency is thus high around the point of zero charge. In addition to pH, which ultimately depends on the concentration of hydrogen ions in the solution, the presence of other ions also affects the collision efficiency. For bare nanoparticles, this mechanism can be predicted by the theory for charging and stabilization as a function of inorganic solution chemistry (ionic composition and pH). This theory is called the DLVO theory (after the names of the creators, Derjaguin, Landau, Verwey and Overbeek) and it describes the electric field just outside a particle's surface towards the solution and how different ionic valences and concentrations in the so called electrical double layer are affecting the extent of this double layer and consequently the stability of the particles (Derjaguin and Landau 1941; Verwey and Overbeek 1948; Elimelech et al. 1995). Divalent ion pairs (such as calcium sulphate) are much more efficient at destabilizing colloidal dispersions since they are more effectively shrinking the extent of the electric double layer than monovalent ion pairs (such as sodium chloride).

At the moment, there exist no established methods for calculation of the collision efficiency. There has been suggestions on how this can be done based on the DLVO theory, such as the following equation from Reerink and Overbeek (1954) with inserts from Elimelech et al. (1995):

$$
\alpha_{i,j} \approx \frac{\sqrt{\dfrac{e^2 \sum c_i z_i^2}{\varepsilon k_{\mathrm{B}} T}}\left(a_i + a_j\right)}{\exp\!\left(\dfrac{\max(V_{\mathrm{A}} + V_{\mathrm{R}})}{k_{\mathrm{B}} T}\right)}
\tag{6}
$$

where c_i is the molar concentration of ion i with a valence of z_i, ε is the permittivity of water, e is the elementary charge, V_{A} is the attractive energy between two particles and V_{R} the repulsive energy. The attractive energy between two particles can be calculated according to:

$$
V_{\mathrm{A}} = -\frac{A a_i a_j}{6 H \left(a_i + a_j\right)}
\tag{7}
$$

where A is the Hamaker constant and H is the distance between the two particles. The repulsive force is described by the following expressions:

$$
V_{\mathrm{R}} = \frac{32 \pi \varepsilon a_i a_j \left(k_{\mathrm{B}} T\right)^2 \tanh^2\!\left(\dfrac{z e \Psi_0}{4 k_{\mathrm{B}} T}\right)}{e^2 z^2 \left(a_i + a_j\right)} \exp\!\left(-\sqrt{\dfrac{e^2 \sum c_i z_i^2}{\varepsilon k_{\mathrm{B}} T}} H\right)
\tag{8}
$$

where Ψ_0 is the surface charge. This parameter is not easy to calculate for the general particle, but for metal oxides it can be calculated according to Dunphy Guzman et al. (2006):

$$
\Psi_0 = \frac{2.303 k_{\mathrm{B}} T}{e}\left(\mathrm{pH}_{\mathrm{pzc}} - \mathrm{pH}\right)
\tag{9}
$$

where $\mathrm{pH}_{\mathrm{pzc}}$ is the point of zero charge. Despite the complexity of these equations, this approach of calculating the collision efficiency has been deemed to be inaccurate in most cases (Wiesner 1992). They do, however, tell that the collision efficiency depend on a number

of additional more basic particle properties, including the Hamaker constant and the point of zero charge. However, these properties do not depend on the particles only, but also on the surrounding medium. Hence, there are currently not sufficient methods to model the collision efficiency from basic particle properties, and thus it not possible to tell exactly which basic particle properties that affect the PEC via the collision efficiency. Consequently, assumptions regarding properties of the medium surrounding the agglomerated particles must be made. Unfortunately, natural waters differ within wide ranges regarding relevant properties, e.g. pH and ionic composition, making it uncertain to make such assumptions.

2.2.3   Natural organic matter

Natural waters contain a large number of molecules and particles in various sizes. The discussion on the DLVO focused on the interaction between nanoparticles and ions. However, natural waters also contain organic matter ranging from very small molecules like organic acids and fatty acids to larger macromolecules like enzymes and polymers like lignin. In natural waters it is well known that sorption of such substances, generally referred to as natural organic matter (NOM), to particles influences the collision efficiency (Arvidsson et al. 2011b). This is because NOM is rich in polyelectrolyte macromolecules with many negative charges, for example humic substances. At most pH values, sorption of NOM provides the nanoparticles with a strong negative charge that can stabilize the particles electrostatically. There are two issues with this mechanism. First, to which extent the sorption of NOM will occur and, consequently, the corresponding change in surface charge is not yet fully theoretically predictable. Second, there is another mechanism of nanoparticle stability that is also believed to be provided by NOM sorption. This mechanism is called steric stabilization, or electrosteric when combined with an electrostatic contribution. Steric stabilization takes place when a hydrophilic polymer or polyelectrolyte is adsorbed to the nanoparticle surface. If then the nanoparticles agglomerate, the water within the hydrophilic polymer layer have to be squeezed out in the process and this would be thermodynamically unfavourable. Stabilization against agglomeration is thus provided. There are no good theoretical descriptions of the electrosteric stabilization of nanoparticles by NOM sorption, and thus no models that can describe how NOM affect the collision efficiency and consequently the PEC of nanoparticles, although it is clear that the effect is considerable and interlinked with pH and ionic strength.

2.2.4   Natural colloids

Agglomeration of particles of the same composition and approximately the same size is referred to as homoagglomeration, while in case of different particle sizes and/or compositions it is referred to as heteroagglomeration. In natural waters, there are typically an abundance of natural colloids (Gallego Urrea et al. 2010). These colloids can be of varying types, such as organic particles and inorganic clay minerals. Considering that the concentration of natural colloids is much higher than the concentration of manufactured nanoparticles in natural waters, heteroagglomeration is probably more environmentally relevant, yet more difficult to model and to study experimentally. Whereas most commercially available nanoparticles have an organic surface coating and thus a negative charge, there are many natural colloids that have a positive charge such as iron and aluminium (hydr)oxides and edges of natural clays. Comparatively few studies of the agglomeration behaviour of manufactured nanoparticles and natural colloids have been conducted. They are thus less well characterized, although the general principles of agglomeration will apply for heteroagglomeration as well. The same problem of determining collision efficiencies for the case of homoagglomeration thus applies for heteroagglomeration as well: It is not known which basic particle properties that control this behaviour and consequently not which basic

particle parameters that affect the PEC of nanoparticles via heteroagglomeration with natural colloids. Furthermore, the same difficulties related to varying properties of natural water are valid for heteroagglomeration as well.

2.2.5 Dissolution

Dissolution of nanoparticles in water has been observed for a few nanoparticle compositions, especially for zinc oxide and silver nanoparticles. Other nanoparticles, such as titanium dioxide and cerium oxide, hardly dissolve at all. Dissolution means that the metal ions are leaving the surface. The presence of counter ions, which either complex or form a new precipitate with the dissolved metal ions, can also accelerate dissolution. In the case of silver nanoparticle dissolution, first the exposed surface of elemental silver needs to be oxidized to silver ions before the ionic form of oxidized silver may leave the surface. The presence of low levels of chloride induces silver chloride precipitation while high levels of chloride such as that found in seawater leads to formation of soluble chloride complexes ($AgCl_2^-$, $AgCl_3^{2-}$, $AgCl_4^{3-}$). In reducing environments, where often significant concentrations of hydrogen sulphide species are present, silver forms silver sulphide precipitates. Silver and nanosilver in wastewater treatment plants has been shown to dissolve or transform into silver sulphide nanoparticles (Kim et al. 2010). In addition to the rates of ions leaving the surface, there is a diffusion of ions and counter-ions into and from the surface through the diffuse boundary layer into the bulk solution. Consequently, there may be an effect of stirring or turbulence on the dissolution rates of nanoparticles. Attempts to derive equations for the dissolution of silver ions from silver nanoparticles have been conducted by Liu and Hurt (2010):

$$\frac{dm}{dt} = - mAe^{-E/RT} \left(\frac{[H^+]}{10^{-7} M} \right) e^{-a[NOM]}$$

(10)

Where m is the mass concentration of silver nanoparticles, the three parameters *E*, *A* and *a* are empirical fitting parameters, [H$^+$] is the concentration of hydrogen ions and [NOM] is the concentration of natural organic matter. As can be seen, this is an empirical equation, and does not provide much guidance on the effect of basic particle properties on the dissolution rate. Thus, which particle properties that affect dissolution and consequently the PEC of nanoparticles are at present largely unknown.

2.2.6 Sedimentation

All particles with a density higher than water will due to gravity have a net downwards force vector, which leads to a certain sedimentation velocity (Elimelech et al. 1995). However, small particles also have a random oriented velocity due to diffusion that is to some extent counteracting the formation of a downward concentration gradient of particles in lakes and seawaters. The diffusion can be either dominated by Brownian motion (self-diffusion) or turbulence depending on the shear flow of the water. Nanoparticles have quite high Brownian motion but in many natural waters the turbulent diffusion is much higher. Due to their small size, non-agglomerated nanoparticles have an almost negligible sedimentation rate, but some exceptions occur for high-density materials such as gold, which can have a substantial sedimentation rate even below 100 nm particle size. Particle sedimentation is well described for particles and agglomerates of spherical dense morphology. However, in the case of nanoparticle agglomerates, these often have a very porous, open structure in natural waters. An equation that can be used to model the change in particle number concentration was used by Arvidsson et al. (2011b):

$$\frac{dn_j}{dt} = -\frac{2a_j^2(r_p - r_w)g j^b n_j}{9\eta d}$$

(11)

where n_j is the particle number concentration of particle j (i.e. if j equals three, then particle j consists of three primary particles that have agglomerated), β is a shape factor that describes the increase in sedimentation rate due to increased cluster size (2/3 for spherical agglomerates (Grant et al. 2001)), and d is the depth of the water. As can be seen, one particle property that affects the sedimentation and thereby the PEC of nanoparticles is again the particle density, as was identified for agglomeration as well. In addition, the shape factor β depends on the shape of the agglomerates, which in turn depend much on properties on the water. If the agglomeration is fast, that is if the value of the collision efficiency is high, the agglomerate shape tends to become more spherical. The shape factor β is thus also connected to the difficult-to-model collision efficiency, and its effect on the PEC of nanoparticles can therefore not easily be determined from basic particle properties.

2.2.7 Nanoparticle coating and aging

As discussed above under the topic of natural organic matter, surface functionalization and modifications play a dominating role in nanoparticle electrosteric stabilization. In addition to becoming coated by natural organic matter in natural water, nanoparticles may already be coated when used in products. According to Arvidsson et al. (2012), most titanium dioxide nanoparticles used in products have coatings of silicon and aluminium oxide and coated nanoparticles may therefore be emitted to the environment. In addition, Kaegi et al. (2008) studied emissions of titanium dioxide nanoparticles embedded in paint applied to façades and found that it was possible to detect coated titanium dioxide nanoparticles emitted from the façade that reached the environment. Exactly how these coatings affect nanoparticle fate is currently unclear, and in order to reveal this it is important that fate studies should be based on the actual nanoparticle formulations rather than the original (uncoated) core material. One study investigated the ageing of nanoparticles and the authors concluded that the ageing of titanium dioxide nano-composites used in sunscreens changed the dispersability in water and consequently the environmental fate (Labille et al. 2010). Coatings, as well as the core material, may also become transformed during aging of particles in the environment. There are several studies that show experimental proof that environmental transformations of the original nanomaterials occur once they are emitted to the environment and after a certain time of aging. One example is the degradation of the polydimethylsiloxane coating on titanium dioxide nanoparticles (Labille et al. 2010), while some more case studies are discussed by Nowack et al. (2012). The effect of different modification and degradation processes on the coating and core material of nanoparticles is an important area for investigation, since there are currently no studies linking specific particle properties to changes in PEC of nanoparticles due to coatings and aging.

2.3 Analysis and conclusions

This section describes that aquatic colloid system with manufactured nanoparticles offers considerable complexity. The fate of nanoparticles in water depend on a number of complex fate mechanisms that are governed by parameters that depend on both properties of the nanoparticles and properties of the environment, in this case natural water. Examples of such parameters include the collision efficiency, the Hamaker constant and the point of zero charge. Consequently, based on the currently available research, it is difficult to find a set of particle properties that can be measured and used in exposure models to calculate a PEC in a

similar fashion as the molecular weight, water solubility, vapour pressure, half-life and the octanol-water partition coefficient outlined in Section 1.3 and Eq. 2. This difficulty of finding general nanoparticle parameters or descriptors that can be used to model the environmental fate of nanoparticles was also noted by Westerhoff and Nowack (2012). The section on emission assessment describes that it is also difficult to obtain accurate data on emissions of nanoparticles as a basis for calculations of the PEC.

3 Review of exposure models of nanoparticles in water

Based on a literature review, the currently existing exposure models for nanoparticles in aquatic environments have been reviewed and compared. The review only includes studies focusing on exposure of nanoparticles. For instance, this means that the study by Blaser et al. (2008), which estimated the total silver exposure and the share of that originating from antibacterial textiles and plastics from which only a share is in the form of nanoparticles, has not been included here. Table 1 lists all included studies, which are also described below, a sub chapter each in alphabetical order according to the first author. After the models have been presented, a comparison is made based on the following aspects: (1) sources of emissions, (2) included basic particle properties in the fate modelling, (3) exposure metrics, (4) materials considered and (5) estimations of PEC values for water. Departing from the analysis of these criteria, an analysis and comparison is conducted, highlighting strengths and weaknesses of the included exposure models.

3.1 The emission model by Arvidsson et al. (2011a, 2012)

This model is aimed at assessing emissions of nanoparticles from the socio-technical system to the environment. Hence, it only includes the emission part of the exposure assessment. The model is developed from the more well-known method of substance flow analysis (SFA) where the flows and stocks of a chemical substance is followed through the raw material acquisition, production, use phase and waste handling (van der Voet 2002). However, particle number is applied as flow and stock metric instead of mass as in traditional SFA. Note also that, opposite to the case of chemical substances, the raw material acquisition is not included in the particle flow analysis (PFA) model since nanoparticles are not extracted from nature but produced from other raw materials. The basic equation describing the flows and stock of a process (production, use phase or waste handling) is:

$$\frac{dN}{dt} = \mathring{a}\, n_{\text{in}} - \mathring{a}\, n_{\text{out}} + n_{\text{s}}$$

(12)

where N represents particle number stock, n particle number flows and the index s stands for sources. The parameter n_s can be a source or a sink depending on its sign, positive (source) or negative (sink). Emissions from the process of interest can then be calculated in different ways depending on the situation. The model is also illustrated graphically in Figure 6.

The sources of emissions included in the PFA model are emissions from production, the use phase and the waste handling of nanoparticles. No fate mechanisms are included and no PEC values are estimated as this model is focused on assessing emissions and do not include the environmental fate. Hence, no particle properties relevant for fate modelling are outlined. The model has so far been applied to titanium dioxide and silver nanoparticles.
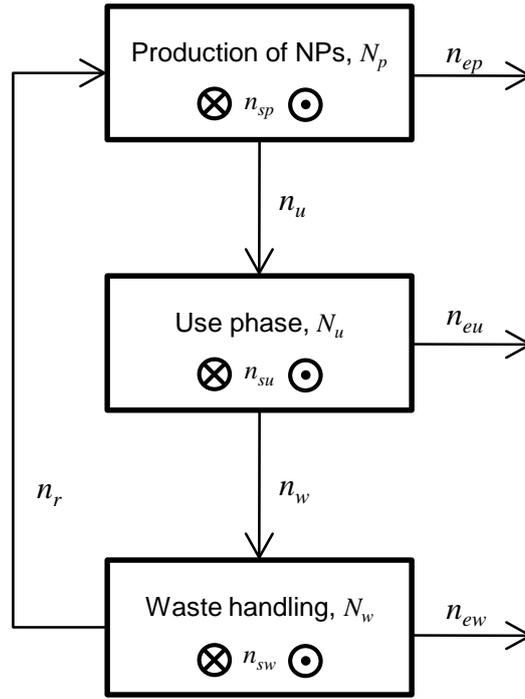
Figure 6. Illustration of the particle flow model. N stands for particle number in stocks, n for particle number, as flows, and the indices p stands for production, u for use phase, w for waste handling, s for source, r for recycling and e for emissions. Sources and sinks due to particle number changes have been represented in the illustration by arrows perpendicular to the others, that is the circles with dots and crosses, respectively. Figure obtained from Arvidsson et al. (2011a).

3.2 Arvidsson et al. (2011b)

This model is based on colloid stability theory and particle collision theory based on the early works by Smoluchowski (1917) and Friedlander (1977) and the more recent work by Grant et al. (2001). The main equation of the model is:

$$\frac{dn_j}{dt} = \frac{1}{2}\sum_{i=1}^{i=j-1} a_{i,j-i}K_{i,j-i}n_i n_{j-i} - n_j\sum_{i=1}^{i=\yen} a_{i,j}K_{i,j}n_i - \frac{v_s}{d}j^b n_j + I_j - n_j\sum_{i=1}^{i=\yen} g_{i,j}H_{i,j}c_i \tag{13}$$

where n_j is the particle number concentration of particle j (i.e. if j equals three, then particle j consists of three primary particles that have agglomerated), $\alpha_{i,j}$ and $\alpha_{i,j-i}$ are collision efficiencies, $K_{i,j}$ and $K_{i,j-i}$ are rate constants, v_s is the sedimentation rate of primary particles, β is a form factor (2/3 for spherical particles), d is the depth of the water compartment, I_j is the inflow of particles, γ is the collision efficiency, H is the rate constant for collisions between natural colloids and manufactured nanoparticles, and c_i is the concentration of an agglomerate with i primary natural colloids. This inflow is the same as the emissions from society. Note that a PEC can be calculated by summarizing dn_j/dt for all values of j. The first term on the right side of the equation describes the formation of particle j through agglomeration of particles i and $j-i$. The second term describes the loss of particle j through agglomeration with other particles i. The third term accounts for the sedimentation, and the forth term for the inflow of particles. The equation is based on the assumption that the particles are spherical. The agglomeration rate constant, which includes three types of agglomeration mechanisms (perikinetic agglomeration, orthokinetic agglomeration and differential settling), can be calculated according to Eq. 14:

$$K_{i,j} = \frac{2k_BT}{3\mu}\frac{(a_i+a_j)^2}{a_ia_j} + \frac{4}{3}G(a_i+a_j)^3 + \left(\frac{2\rho g}{9\mu}\right)(\rho_p - \rho_w)(a_i+a_j)^3(a_i - a_j) \tag{14}$$

where k_B is the Boltzmann constant, T is the temperature, μ is the fluid viscosity, a_i and a_j are the radii of particles i and j respectively, G is the shear rate, g is the gravitational acceleration and ρ_p and ρ_w are the densities of the particles and water, respectively. The sedimentation rate in the main equation is calculated according to Grant et al. (2001) based on Stoke's law:

$$v_s = \frac{2a_j^2(\rho_p - \rho_w)g}{9\mu} \tag{15}$$

where a is the radius of the primary particle. As should be clear by now, this model focuses on agglomeration and sedimentation. The effects of natural organic matter, which have been highlighted as important in many experimental studies (Arvidsson et al. 2011b), can be included indirectly via the collision efficiency if such experimental values are available. The exposure metrics used is particle number rather than mass. The emissions are included in the model as a generic inflow (I). In Arvidsson et al. (2011b) titanium dioxide nanoparticles was used as a case for performing the calculations. However, no explicit PEC was estimated. The particle properties that are included in the fate modelling are particle size and density. In addition to these two, the collision efficiency is an included parameter that depends partly on properties of the particles but also of properties of the water.

3.3 Boxall et al. (2007)

In a report by Boxall et al. (2007) the following model for prediction of PEC for surface water is presented:

$$\text{PEC} = \frac{m_{\text{direct}}}{V} + \frac{m_{\text{soil}}\left(f_{\text{spray drift}} + f_{\text{runoff}}\right)}{V} + \frac{CUF\left(1 - f_{\text{stp}}\right)}{Wd} \tag{16}$$

The first of the three terms is the direct application of nanoparticles to surface water, the second term is the inflow of nanoparticles from soil, which includes emissions from spray drift and runoff and assumes that each hectare of agricultural field is linked to a 35 thousand litre stream, and the third term is the inflow to the surface water from sewage treatment plants. m_{direct} is the direct application rate of nanoparticles to surface water, V is volume of the water body, m_{soil} is the application rate of nanoparticles to soil, $f_{\text{spraydrift}}$ is the fraction of nanoparticles released in spray drift, f_{runoff} is the fraction of the nanoparticles applied to soil that enter the water via runoff, C is the concentration of nanoparticles in products, U is the daily usage of products, F is the market penetration of the product, f_{stp} is the fraction of nanoparticles removed in sewage treatment plants, d is a dilution factor and W is the amount of waste water produced.

As can be seen from the equation, no fate mechanisms are included in this model. It is simply assumed that all nanoparticles emitted to the water are distributed evenly in the water volume. Thus no specific physiochemical nanoparticle parameters are included. However, the model is very explicit compared to other models as to where the nanoparticles originate from since it includes three distinct sources. The exposure metrics is mass and the model includes four sources, which are direct application, spray drift, runoff from soil, and inflow from sewage treatment plants. The nanoparticulate materials for which the model has been applied are

silver, aluminium oxide, gold, cerium oxide, fullerenes, hydroxyapatite, latex, organosilica, silicon dioxide, titanium dioxide and zinc oxide. Estimated PEC for water from his model can be found in Table 1.

3.4 Gottschalk et al. (2009)

The model presented by Gottschalk et al. (2009) in a number of articles is a development of the early model by Müller and Nowack (2008). It is a mass flow model primarily intended to be used on a national or regional scale. The part of the model, which describes the concentration of nanoparticles in surface water, can be described mathematically as:

$$\text{PEC} = \frac{m_{stp} + m_{stp,untreated} + m_{pmc} + m_{soil} + m_{atm} + m_{sed1} - m_{sed2} - m_{gw} - m_{out}}{V} \quad (17)$$

where m_{stp} is the flow of nanoparticles from sewage treatment plants to the surface water via the treated water, $m_{stp,untreated}$ is the flow of nanoparticles from sewage treatment plants to the surface water via the untreated water, m_{pmc} is the direct flow of nanoparticles from production, manufacturing and consumption of nanoparticles in society to the surface water, m_{soil} is the flow of nanoparticles from the soil compartment to the surface water, m_{atm} is the flow of nanoparticles from the atmosphere to the surface water, m_{sed1} is the flow of nanoparticles from the sediments to the surface water, m_{sed2} is the flow of nanoparticles from the surface water to the sediment, m_{gw} is the flow of nanoparticles from the surface water to the ground water, m_{out} is the flow of nanoparticles which leaves the surface water and exits the system boundaries (typically the borders of the country or region of interest), and V is the volume of the surface water. It is somewhat difficult to tell which fate mechanisms that are included in this model. Since the mass flow of nanoparticles from the surface water to the sediment is included one may say that the fate mechanism of sedimentation is included. But the mechanism is not at all described in the article in which the model is presented. It is, however, clear that no specific physiochemical properties of the nanoparticles are included and that mass is applied as exposure metric. The included four sources of emissions are sewage treatment plants, consumption and production, soil and air. The calculations were performed for titanium dioxide, zinc oxide, carbon nanotubes and fullerenes. Estimated mode values of PEC for Europe from this model can be found in Table 1.

3.5 Gottschalk et al. (2011)

In this model, environmental concentrations of titanium dioxide, zinc oxide and silver nanoparticles are estimated using the same emission data, and consequently the same emissions sources, as in Gottschalk et al. (2009). Again, mass is used as exposure metric. However, in this study the focus is more clearly on the aquatic environment, in particular Swiss rivers, with the aim of introducing spatial and temporal resolution into the model. The annual emissions of nanoparticles in Switzerland, obtained by assuming that the global production is distributed according to population density, were distributed across the country proportional to population density and minimal dilution was assumed. Complete mixing of sewage water and river water was assumed. Two scenarios were applied for the transport of nanoparticles in rivers: one denoted S_0 with complete nanoparticle removal between two catchments inflow points and one denoted S_c with no removal at all between two catchments. These scenarios were suggested to cover all possible fate scenarios for the nanoparticles. Hence, it is again difficult to tell which fate mechanisms are included in this model, but it is clear that no physiochemical nanoparticle properties are included in the modelling. The

modelling was conducted for 21 river sections in Switzerland and the results show considerable variation in time and space. Considering that the results varied in both time and space and for the two scenarios S_0 and S_c no specific PEC values can be reported in Table 1 for this model. Instead we recommend the reader to read the original article in order to explore the ranges of concentrations for the different scenarios.

3.6 Johnson et al. (2011)

In their paper, Johnson et al. (2011) estimate environmental concentration of titanium dioxide nanoparticles from sunscreen in the river Thames in the United Kingdom. Hence, the use of the product sunscreen is the only source considered and titanium dioxide nanoparticles is the only material considered. PEC values were calculated using the Low Flows 2000 Water Quality extension (LF2000-WQX) model, which is a model used for studying release of contaminants from consumer products. This model has previously mainly been used for organic contaminants, see for instance Williams et al. (2009). It combines hydrological models estimating magnitude of and variations in flows across a compartment with water quality models. The fate mechanisms included are thus the same as those for chemicals in the LF2000-WQX model, which are removal during wastewater treatment, dilution and biodegradation. Consequently, none of the specific fate mechanisms pertaining to nanoparticles are included and thus no physiochemical nanoparticle properties either. Mass is used as exposure metric and estimated PEC values can be found in Table 1.

3.7 Musee (2011)

In this model, the PECs of nano-sized silver and titanium dioxide from cosmetics were modelled for the case of Johannesburg city, South Africa. The main model equation was

$$\text{PEC} = \frac{10^{12} m}{PWD} \tag{18}$$

where m is the mass input of the nanoparticle to the water, P is the relevant population (Johannesburg city in this case), W is the wastewater volume per capita and D is a dilution factor in the receiving water. According to Musee (2011), this approach is similar to the procedure for chemical risk assessment recommended by the European Chemicals Bureau, which is now replaced by the European Chemical Agency (ECHA). The input of nanoparticles was estimated based on estimations of the same parameter for Switzerland, but correlated to fit the wealth, population and market penetration of nanoparticle-containing cosmetics of South Africa.

This model is similar to the third term in the model by Boxall et al. (2007) which also focused on emissions of nanoparticles which pass the sewage treatment plant. In this model as well, no direct fate mechanisms are included, no specific physiochemical properties of the particles are included either, and mass is applied as exposure metric. The model considers two sources, nanoparticles from untreated water, which is from production and consumption, and nanoparticles that were not removed in sewage treatment, which makes sewage treatment the second source. The model is applied for silver and titanium dioxide, see Table 1 for PEC values for their baseline case. The values are approximate since Musee (2011) applied a number of different dilution factors, thereby receiving somewhat different PEC values.

3.8 O'Brien and Cummins (2010)

This study investigated the exposure of titanium dioxide, silver and cerium dioxide nanoparticle from consumer products to aquatic environments in Ireland. The product considered for titanium dioxide nanoparticle release was exterior paint, for silver nanoparticle release the considered product was food packaging, and for cerium dioxide nanoparticle release it was as additive in diesel fuel. The model is based on the following equation:

$$\text{PEC} = \frac{fm + m_{\text{direct}} + m_{\text{air}} + m_{\text{leach}}}{V} \quad (19)$$

where m is the nanoparticle release to surface water from waste water treatment plants, f is the fraction of treated wastewater released to freshwaters in Ireland, m_{direct} is the direct release to surface water from the products, m_{air} is the deposition, m_{leach} is the leakage of nanoparticles from soil and V is the volume of the water. The study focuses mainly on assessing the removal of nanoparticles in wastewater treatment plants and no fate mechanisms in the water are included. Consequently, no physiochemical properties of the nanoparticles are included. Mass is used as exposure metric and estimated mean values of PEC can be found in Table 1.

3.9 Praetorius et al. (2012)

In the study by (Praetorius et al. 2012), PEC values for the Rhine river are estimated based on a river model combined with a model for nanoparticle environmental fate. The stressor studied is titanium dioxide nanoparticles. The model is much based on kinetic fate mechanisms for particles, as described in Section 2.2. The fate mechanisms of agglomeration, sedimentation and advection are included. However, homoagglomeration (see Section 2.2.1) is not included as the probability of two manufactured nanoparticles colliding in the environment is considered unlikely considering the low concentration of manufactured nanoparticles compared to the concentration of natural colloids. Instead, the study focuses on heteroagglomeration with natural colloids (see Section 2.2.4). Considering the difficulties of modelling collision efficiencies, different scenarios for its value (1, 0.1, 0.01 and 0.001) are applied. The emissions assumed were the same as in Gottschalk et al. (2011), which includes a number of sources in Switzerland (production and consumption, sewage treatment plants, air and soil).

This study is unique since it estimates PEC values both in terms of particle number and in terms of mass. Mass-based emission data is first recalculated into particle number, since many particle fate processes depend on particle number rather than mass (see Section 2.2). The resulting particle-based PEC values are then recalculated into mass-based PEC-values as well, and both are presented. In addition, the PEC values are estimated as a function of the distance to the source. The maximum PEC values, obtained close to the source, are about $3.3 \cdot 10^8$ particles/m^3 or, in terms of mass, about 5.5 ng/L.

The basic particle properties included in the model are particle size and density. In addition, the fractal dimension (D_f) of the agglomerates is included in the model. This is a parameter that is similar to the form factor β in the model by Arvidsson et al. (2011b), since they are both dimensionless and provide information about the shape of the agglomerates. However, similar to the form factor β, the fractal dimension is not a particle property only, but also depends on properties of the water. For example, if the water contains high concentration of multivalent ions, the agglomeration becomes fast and then the agglomerates will become spherical and have a lower fractal dimension. In addition to these parameters, the collision efficiency is also included in this model.

3.10 Quik et al. (2011)

The mass-based model developed by Quik et al. (2011) can be formulated as:

$$\frac{dc}{dt} = m - \left(k_{adv} + k_{sed} + k_{diss}\right)c \tag{20}$$

where c is the mass concentration of nanoparticles, t is the time, m is the emissions to the aquatic system, k_{adv} is the rate constant for advection, k_{sed} is the rate of sedimentation and k_{diss} is the rate constant of dissolution. Quik et al. (2011) suggest ranges of values for these rate constants in their article which typically vary by two or three orders of magnitude. No attempt to calculate the rate constants from basic physiochemical properties for a general case is done in the study. Hence, these three parameters are problematic in a similar manner as the collision efficiency in that they depend on both particle and water properties in a complex manner.

The model by Quik et al. (2011) was developed based on how exposure of chemical substances is calculated for aquatic organisms. It includes three fate mechanisms as indicated by the model equation. First, advection, which is when the nanoparticles are transported away by a water stream such as a river departing from a lake of interest, is included. Second, sedimentation, which is when the nanoparticles leave the water body and enter the sediment due to gravitation, is included. Here, they acknowledge that agglomeration plays an important role in the sedimentation of nanoparticles mostly because the nanoparticles agglomerate with natural colloids which eventually sediment. The fate mechanism of agglomeration is hence implicitly included in the sedimentation process. Third, dissolution, which is when the nanoparticles dissolve into ions or molecules, is included. Some nanoparticles, such as silver and zinc oxide, dissolve easily whereas titanium dioxide nanoparticles hardly undergo any dissolution at all in natural waters. Mass is applied as exposure metric in the model. This model includes a generic term for emissions (E). No PEC values were reported.

3.11 Analysis and conclusions

Table 1 shows (1) sources of emissions, (2) included basic particle properties in the fate modelling, (3) exposure metrics, (4) materials considered and (5) estimations of PEC values for water for the included studies.

Regarding emissions, it can be seen in Table 1 that some models include a generic term for emissions and do not further elaborate on their origin but focus on the fate of the particles in the water. Other models elaborate more on the sources of the emissions. Sewage treatment plants are commonly included sources of emissions, along with direct emissions from production and consumption in society. The most detailed modelling of emissions of nanoparticles from society is done in the PFA studies by Arvidsson et al. (2011a, 2012).

Regarding basic particle properties, it can be seen from Table 1 that there are only two studies so far that relate specific physiochemical particle properties to the PEC, namely Arvidsson et al. (2011b) and Praetorius et al. (2012). The other studies rely more on mass flow modelling of environmental fate. This points to the merits of the former two studies since such approaches would enable generic exposure models based on measurable particle properties in a similar manner as is currently done for non-nanoparticulate chemicals.

Regarding exposure metric, of the studied studies conducted so far (autumn 2012), the only ones exploring other exposure metrics than mass are Arvidsson et al. (2011a; 2011b; 2012) and Praetorius et al. (2012). Considering the extensive discussion that mass may not be an

adequate exposure metric for nanoparticles (Handy et al. 2008; Abbott and Maynard 2010) it is unfortunate that no other studies explore other metrics. Still, considering that the vast majority of the toxicological and ecotoxicological studies on nanoparticles are conducted on mass-basis (Baun et al. 2008; Kahru and Dubourguier 2010), apart from some few which advocates other metrics (Oberdörster et al. 2005; van Hoecke et al. 2008), it may be difficult to use results from exposure assessments with other metrics than mass for calculating risk ratios. If other metrics then mass are to be used in risk assessment of nanoparticles, it requires efforts from both exposure assessors and (eco)toxicologists. It is also notable in this context that the definition suggested by the European Commission is based on particle number, see Section 1.1.

Regarding materials considered, it is worth noting that the studies include a wide range of nanoparticles with regards to chemical composition, although titanium dioxide and silver are studied more often than the others. This is not surprising since silver is the most frequently found nanomaterial in consumer products, and titanium the third most frequently found (Project on Emerging Nanotechnologies 2012).

Regarding PEC values, it can be seen in Table 1 that the numerical PEC values estimated in the included studies vary several orders of magnitude. This is not surprising since the studies include different sources of emissions and are applied for different geographical areas. The lack of comparison to measured concentration of the included nanoparticles also makes it difficult to analyse the accuracy of these numbers.

Finally, Quik et al. (2011) discusses the order of kinetics extensively. They argue that although agglomeration is a process following second order kinetics, as described by Arvidsson et al. (2011b), it is possible that the majority of the particle collisions involving nanoparticles in the environment takes place between a manufactured nanoparticle and a natural colloid. Since the concentration of natural colloids is so much higher it can be seen as almost constant and then the agglomeration will only depend on the concentration of manufactured nanoparticles in a first order kinetic way. However, this simplification needs to be further motivated. A common sense-reflection may be that there is a large difference between a large river (e.g. the river Rhine) and a clear mountain stream with regards to concentration of natural colloids. Consequently, there is need to evaluate when a first order kinetic approximation may be valid and when more detailed second order kinetics should be applied. In general, there is always a need to validate developed exposure models, and how this could be done for the case of nanoparticles is further discussed below.

Studies	Sources of emissions	Basic particle properties in fate modelling	Exposure metrics	Materials considered	PEC estimated [µg/l]
Arvidsson et al. (2011a, 2012)	Production, use phase, waste handling	None	Particle number	TiO$_2$ Ag	-
Arvidsson et al. (2011b)	Generic	Particle size Particle density	Particle number	TiO$_2$	-
Boxall et al. (2007)	Direct application Spray drift Runoff from soil Sewage treatment plants.	None	Mass	Ag Al$_2$O$_3$ Au CeO$_2$ Fullerenes Hydroxyapatite Latex Organo-SiO$_2$ SiO$_2$ TiO$_2$ ZnO	0.01 0.0002 0.14 <0.0001 0.31 10 100 0.0005 0.0007 25 76
Gottschalk et al. (2009)	Production and consumption Sewage treatment plants Air Soil	None	Mass	TiO$_2$ ZnO Ag CNT Fullerenes	0.015 0.010 0.764 0.004 0.017
Gottschalk et al. (2011)	Production and consumption Sewage treatment plants Air Soil	None	Mass	TiO$_2$ ZnO Ag	Different ranges of values reported, see further the reference.
Johnson et al. (2011)	Use of sunscreen	None	Mass	TiO$_2$	<8.8
Musee (2011)	Production and consumption Sewage treatment plants	None	Mass	Ag TiO$_2$	~0.001 ~0.001
O'Brien and Cummins (2010)	Use phases of exterior paint, food packaging and diesel fuel	None	Mass	Ag CeO$_2$ TiO$_2$	0.029 0.024 1.45
Praetorius et al. (2012)	Production and consumption Sewage treatment plants Air Soil	Particle size Particle density	Particle number and mass	TiO$_2$	<3.3·10^8 particles/m3 or <5.5 ng/L
Quik et al. (2011)	Generic	None	Mass	-	-

Table 1. Review of five exposure models for nanoparticles in aquatic environments. Abbreviations: Titanium dioxide nanoparticles (TiO$_2$), silver nanoparticles (Ag), aluminum oxide nanoparticles (Al$_2$O$_3$), gold nanoparticles (Au), cerium oxide nanoparticles (CeO$_2$), silica/silicon dioxide nanoparticles (SiO$_2$), zinc oxide nanoparticles (ZnO) and carbon nanotubes (CNT).

4 Considerations for experimental validation

Few of the studies reviewed in Chapter 3 attempt to validate their results. Gottschalk et al. (2009) compared their results of concentrations of titanium dioxide nanoparticles in sewage treatment plant effluents to measured concentrations in the same media from one study, and found that the numbers were in the same range. However, their modelled titanium dioxide nanoparticle concentration was compared to measured total titanium smaller than 0.7 µm, which would contain other forms of colloidal titanium.

The validation of exposure models can be done in sequential steps by quantitative parameterization of individual fate processes such as those described in Chapter 2. Another approach to experimentally validate exposure models of nanoparticles in environment is to compare the output result of a model to experimental measurements. The media used in the experimental setup can have typical characteristics of a particular environment relevant to investigate or even be a real natural sample. This more aggregated approach has the advantage that it is possible to identify concomitant effects of different fate processes that could become overlooked with the individual-process evaluation approach. The considerations mentioned here are, however, applicable for any of the two fate validation approaches mentioned above.

Here we focus on agglomeration processes in the same manner as they are described in Chapter 2. As was shown, there are a number of physicochemical properties that influence the degree of agglomeration in different ways. As a first example we can take the simplest form to measure agglomeration of nanoparticles, which is homoagglomeration in the presence of a monovalent counter-ion only. This means that no natural organic matter or natural colloids are present, leaving out many of the fate mechanisms shown in Figure 4. In this case, a number of critical factors can be identified for experimental validation. The first of these is the origin of the surface charge (Hiemenz and Rajagopalan 1997). Surface charges can originate from pH-dependent hydroxylation of oxide surface groups (e.g. for metal oxides), from preferential adsorption of ions (e.g. I^- for AgI particles), dissociation of surface groups (e.g. for latex particles), isomorphic substitution (e.g. for clays), adsorption of polyelectrolytes or accumulation of electrons (e.g. for metal particles). If the surface charge is strongly dependent on pH, as for the case of metal oxides, then the pH value is going to be critical for the final outcome of the experiment. If the surface charge is determined by any other mechanism, then the nature of the ions needs to be carefully considered in order to avoid the use of potentially determining ions. The second critical factor is hydrodynamic interactions. The degree of mixing needs to be controlled in order to warrant homogeneous velocity gradients in the experiments that generate very similar conditions leading to agglomerates that are comparable with other replicate experiments. The mixing is controlled by the shear rate G in Eq. 4. The third critical factor is the container material. This factor is of paramount importance because the induction of agglomeration by addition of an electrolyte is based on the screening of the electrical double layer surrounding the particles and a collateral effect is that the particles may undergo deposition to the container walls. This is especially likely if the nanoparticles and the container material attract each other. Container material must thus be chosen with great care.

The three factors mentioned above are even more important when performing experiments with natural waters as the effects of ions, organic matter, pH change and hydrodynamic factors will all influence the final outcome and a small difference in initial mixing of the suspension can result in a different particle size distribution or one process can be favoured over another. The experimental outcome is thus very sensitive to variations in initial conditions. This is also true for the actual case of an eventual release to the environment and therefore it is important to characterize the actual form of release and state of the

nanoparticles when released into the environment. Although this insight is derived from experiments on homoagglomeration, the same principle is likely applicable for heteroagglomeration as well.

It is also important to use an appropriate number of particles when studying agglomeration behaviour of particles in an experimental setup. The determination of the appropriate number of particles can be inferred from the first two terms in Eq. 3. The particle number concentration will impact the rate with which particles will agglomerate; increasing the number of particles per volume will increase the probability of encountering another particle. The initial particle number concentration will thus have a considerable impact on the experimental outcome. In addition, some researchers have found that the surface charge of nanoparticles is dependent on particle size (Abbas et al. 2008). This might lead to higher energy barriers at smaller particle sizes and consequently lower agglomeration rates.

Furthermore, the particle size may constrain which experimental methods that can be utilized. One common method for studying agglomeration behaviour and measure particle size is dynamic light scattering (DLS) (Hasellöv and Kaegi 2009). One output of a DLS experiment with regards to agglomeration behaviour is the particle number concentration half-time, which is the time it takes to reduce the particle number by 50%. DLS measures the intensity fluctuations of scattered light by an ensemble of particles and the minimum number of particles required for producing results increases with decreasing particle size due to the detection limitations of the light sensor. Consequently, and as indicated above, if the particles are small they agglomerate fast and the agglomeration cannot be monitored by DLS. The minimum particle size that is detectable in DLS can be calculated in advance. In a DLS instrument called the Malvent zetasizer-ZS it was found that the minimum concentration measurable for an initial particle size of 15 nm was 100 mg/l of titanium dioxide nanoparticles. Table 2 gives an example for half-life times reaction values for this concentration of titanium dioxide nanoparticles using the equation for half-life times from Petosa et al. (2010) and assuming homogeneous particle size distribution. For particles in the range of 10 nm, the halftime becomes too small for the concentrations required by DLS. For particles sizes larger than 50-60 nm, however, DLS techniques are able to measure the concentrations required to obtain approximate agglomeration rates based on particle size measurements.

In addition to DLS, nanoparticle tracking analysis (NTA) can also be used for measuring the agglomeration rates of nanoparticles. However, there is an inconvenience regarding detection limits for this instrument as well. As stated by Gallego-Urrea et al. (2011), the lowest limit in terms of size is dependent on the refractive index of the particles (an indicator of their ability to interact with light) and, therefore, the lowest initial particle size that can be detected is limited by the refractive index of the particles.

Particle mass concentration [mg/L]	Particle diameter [nm]	Half-time [s]
100	10	$3.9 \cdot 10^{-3}$
100	20	$3.1 \cdot 10^{-2}$
100	50	$4.9 \cdot 10^{-1}$
100	100	$3.9 \cdot 10^{0}$
100	200	$3.1 \cdot 10^{1}$
100	400	$2.5 \cdot 10^{2}$

Table 2. Time required for formation of a dimer from two primary particles, assuming homogenous starting particle size and spherical particles of titanium dioxide. Calculations based on Petosa et al. (2010).

5 Conclusions and recommendations

Knowledge of the emissions of nanoparticles to the environment is crucial input for estimating PEC of nanoparticles, as discussed in Chapter 2. Most of the studies reviewed in Chapter 3 report that better data on emissions of nanoparticles would be beneficial for the exposure assessment. The most urgent action is then to assess the amount of nanoparticles used in society. As discussed in Section 2.1, this information is often lacking. Considering this, *we recommend that information of flows and stocks of nanoparticles in society should be gathered.*

The second important part of an emission assessment is to have an emission factor. There exist no models to aid such estimations, or even provide the correct order of magnitude. Studies from which emission factors may be derived currently only exist for a limited number of nanoparticles and uses, such as silver nanomaterials in consumer products (Benn and Westerhoff 2008; Geranio et al. 2009; Benn et al. 2010; Kaegi et al. 2010; Swedish Chemicals Agency 2011; Lorenz et al. 2012), titanium dioxide nanoparticles in paint (Kaegi et al. 2008) and nanopowder coatings (Hsu and Chein 2007) and carbon nanotubes from composites (Movahedi 2008). Considering the crucial importance of emission factors for assessing emissions and exposure, *our recommendation is that emission factors should be determined and reported for each product that makes use of nanoparticles.* It is still unclear if the emissions should be best reported in terms of mass or particles, since different studies apply different metrics. Until this question is resolved, *the most reasonable recommendation for now would be to report emissions in terms of both mass and particle number.*

In Chapter 2, it was also shown that there are a number of fate mechanisms relevant for nanoparticles that are not accounted for in the exposure assessment methods for ordinary chemicals as conducted, for instance, within the chemicals legislation REACH. Examples include agglomeration, dissociation and reactions with natural organic matter. However, despite being important for nanoparticle fate, most of these fate mechanisms are still difficult to include in an exposure model for deriving PEC values. For non-nanoparticulate chemicals, the impact of different fate mechanisms on the PEC can be modelled based on some basic chemical properties, such as the water-octanol partitioning coefficient, half-life, molecular weight and vapour pressure. Doing the same for nanoparticles appears not to be currently possible. Therefore, in line with Westerhoff and Nowack (2012), *we recommend more research for development of exposure and fate models based on generic particle properties.*

Preferably, this should be done by collaborative efforts between experimental work and modelling in order to ensure both a sound natural science basis and compatibility to generic exposure models (Praetorius et al. 2013). However, it seems that particle size and density constitute important particle or particle-water properties that must be known in order to enable an exposure assessment of a nanoparticulate material. *We therefore recommend that particle size, particle size distribution and density should be reported for nanoparticles on the market in order to enable exposure assessments.*

In Chapter 3, it was shown that most conducted exposure assessment studies on nanoparticles did not include any nanoparticle-specific properties in their models. Although mass flow models, such as Gottschalk et al. (2010) and Musee (2011), may give indications about potential risks of specific nanoparticles, they offer no generic method to assess exposure of nanoparticles based on generic particle properties. For example, Wiesner et al. (2009) commented on the work of Gottschalk et al. (2009) that "such simple models are entirely appropriate for poorly characterized systems, but they offer limited guidance." However, in those models that do attempt to calculate PEC values based on basic particle properties, that is the models by Arvidsson et al. (2011b) and Praetorius et al. (2012), only some fate mechanisms could be included due to the difficulties discussed in Section 2.2. This takes us back to the recommendation above about more research needed to develop exposure and fate models based on generic particle properties.

It is, however, not certain that all fate mechanisms discussed in Chapter 2 are required in order to estimate reasonable PEC values for nanoparticles. In order to investigate when a model is sufficiently accurate, it must be validated by experimental research. How this can be done, and which considerations that need to be taken into account, was briefly discussed in Chapter 4. The general conclusion was that although experimental validation is important, currently there are a number of considerable limitations of the experimental methods that need to be considered. Consequently, *we recommend that more research is needed for overcoming limitations in experimental measurements of nanoparticles for validation of exposure and fate models.*

In general, we have to conclude that there is a mismatch between the need for scientifically well-founded models from risk assessors and authorities and what can currently be provided in terms of available, validated and ready-to-use reliable models from scientists.

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