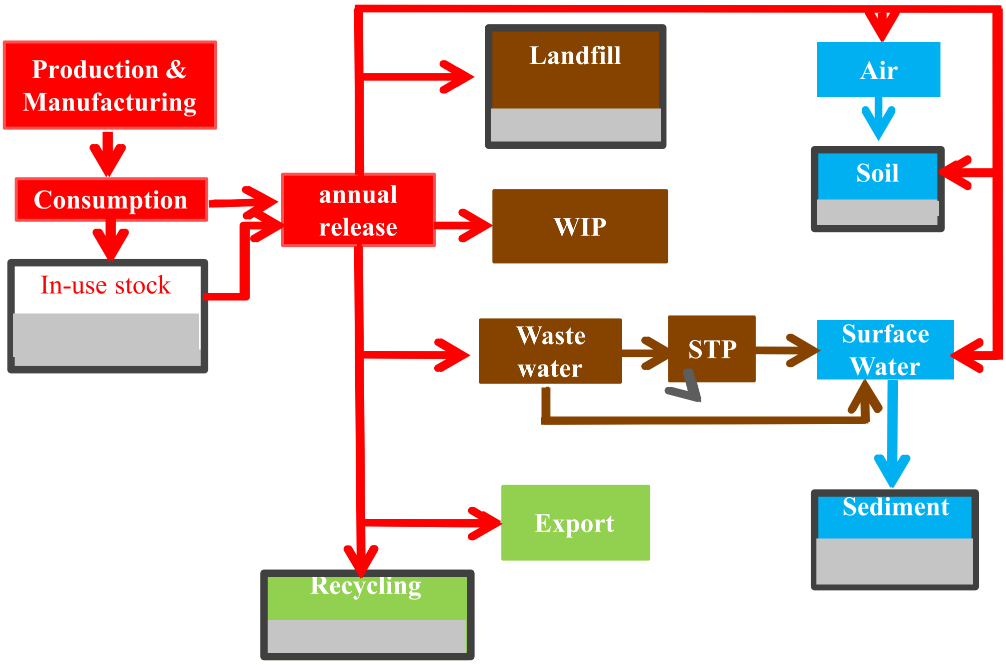
Empa
Materials Science and Technology

Final Project Report

Probabilistic modeling of environmental concentrations and risks of engineered nanomaterials (ENMs)



Im Auftrag des Bundesamtes für Umwelt (BAFU)

St. Gallen, 28. November 2017

Impressum

Auftraggeber: Bundesamt für Umwelt (BAFU), Abt. Wasser, CH-3003 Bern

Auftragnehmer: EMPA- Swiss Federal Laboratories for Materials Science and Technology
Environmental Risk Assessment and Management Group

Autoren: Yan Wang, Prof. Dr. Bernd Nowack
EMPA- Swiss Federal Laboratories for Materials Science and Technology
Environmental Risk Assessment and Management Group
Lerchenfelstrasse 5
CH-9014 St. Gallen

Datum: 28.11. 2017

Begleitung BAFU: Dr. Saskia Zimmermann-Steffens

1. Content

2. *Introduction*

Due to the ubiquity of nanoapplications in our society, the presence of engineered nanomaterials (ENM) in the environment is inevitable, making environmental risk assessments for ENMs imperative. The standard risk assessment approach is to compare the exposure with the toxicity. The first step in environmental risk assessment is therefore to acquire knowledge regarding exposure of ENMs. Currently, techniques for detecting and quantifying ENMs in natural systems are still scarce. Thus, a modelling approach is the best alternative to obtain predicted environmental concentrations (PEC). Several environmental exposure models have been developed during the last decade. Of those available models, the probabilistic MFA (PMFA) models are able to provide mass flows of ENMs towards the environment through an in-depth analysis of the whole system, considering the complete range of nanoproducts. Moreover, caused by the scarcity of data regarding production, use and release of ENMs, the input data for the models is associated with high uncertainties. The PMFA model is able to handle the inherent uncertainty by introducing each input parameter as a probabilistic distribution. As a latest improvement of the PMFA, the dynamic probabilistic MFA model (DPMFA) was developed, considering a dynamic input and release of ENMs. The PMFA and DPMFA model have previously been applied to estimate environmental flows and concentration of a range of ENMs.

The second step in the risk assessment process is to estimate the hazard effects posed by ENMs. A large number of studies have investigated the adverse effects of ENMs. However, ecotoxicological concentrations vary a lot among studies. The species sensitivity distribution (SSD) method is able to consider all available data to calculate the predicted no observed effect concentration (PNEC). A handful of environmental hazard assessment studies for ENMs have been conducted using the SSD method. Among the studied ENMs, nano-TiO_2, nano-ZnO, nano-Ag, and CNTs are the most investigated ones.

The risk characterization ratio (RCR), obtained by dividing the PEC by the PNEC value, describes the potential of risk - a RCR below one means that the risk is under control. If the RCR is above one, the risk needs to be evaluated more carefully and risk management measure may be required to be taken. Only few studies have so far combined exposure and hazard data and performed a holistic environmental risk assessment.

Besides the well-studies ENM mentioned above, little is known regarding the environmental exposure and risk of other ENM, for example nano-$SiO_{2,}$ nano iron oxides, nano-$CeO_{2,}$ nano-$Al_2O_{3,}$ and quantum dots. The aim of this project was therefore to quantify the release of these five ENMs from nanoproducts to the environment using the PMFA and DPMFA models and to conduct an environmental risk assessments.

3. *Main Results*

2.1 Static modelling

In the first part of this project risk assessments were conducted for nano-SiO_2 and nano iron oxides for the EU and Switzerland. The mass flows of these two ENMs were modelled and PECs were obtained for air, soil, surface water, and sediments using a PMFA model. The highest concentration of nano-SiO_2 was found in sediment in the complete sedimentation scenario, with a median annual increase of 0.43 mg/kg·y in the EU (0.19–12 mg/kg·y, 15/85% quantiles). Moreover, the concentration in surface water was predicted to be 0.12 µg/l in the EU (0.053–3.3 µg/l, 15/85% quantiles). The PEC values for nano iron oxides are much lower. The median concentration in surface water was predicted to be 3.4 ng/l in the EU (0.011–36 ng/l, 15/85% quantiles), and the yearly concentration increase in sediment is predicted to be 390 µg/kg·y (0.16-490 µg/kg.y, 15/85% quantiles). The concentrations for these two ENMs in other compartments are shown in Table 1 and Table 2.

Table 1. Mean, mode, median values and 15th and 85th percentiles of predicted nano-silica concentrations (or yearly increase in concentration) in different environmental and technical compartments

			EU			
Compartment	**mean**	**mode**	**median**	**Q15**	**Q85**	**Unit**
Soil	6.2	0.42	0.43	0.14	11	µg/kg.y
Sludge treated soil	6,300	370	420	160	11,000	µg/kg.y
Surface water	1.9	0.037	0.12	0.053	3.3	µg/L
Sediment	6.7	0.13	0.43	0.19	12	mg/kg.y
Air	1.9	0.07	0.12	0.0029	3.5	ng/m^3
STP Effluent	56	1.3	3.8	0.61	83	µg/L
STP sludge	940	36	63	25	1,600	mg/kg
Solid waste	160	2.9	10	4.4	280	mg/kg
WIP bottom ash	1,000	12	68	28	1,800	mg/kg
WIP fly ash	1,400	61	93	37	2,400	mg/kg
			CH			
Compartment	**mean**	**mode**	**median**	**Q15**	**Q85**	**Unit**
Soil	21	0.64	1.3	0.42	34	µg/kg.y
Sludge treated soil	--	--	--	--	--	µg/kg.y
Surface water	2.8	0.19	0.17	0.064	4.4	µg/L
Sediment	9.8	0.66	0.60	0.23	16	mg/kg.y
Air	5.6	0.21	0.3	0.00015	9.5	ng/m^3
STP Effluent	900	43	57	8.7	1,200	µg/L
STP sludge	1,400	83	86	35	2,300	mg/kg
Solid waste	75	1.6	4.5	1.9	120	mg/kg
WIP bottom ash	920	23	54	24	1,500	mg/kg
WIP fly ash	1,000	77	63	26	1,700	mg/kg

Table 2. Concentrations of nano-FeOX in technical and environmental compartments. For soils and sediments, the yearly increase in concentration is given. All values are rounded to two significant numbers.

			EU		
	Q15	**Mean**	**Median**	**Q85**	**Unit**
Air	1.1E-04	2.9E-01	3.4E-02	3.6E-01	ng/m3
Sediment	1.6E-01	390	44	490	µg/kg·y
N&U* soil	3.8E-05	9.4E-02	1.1E-02	1.2E-01	µg/kg·y
ST* soil	4.4E-02	110	13	140	µg/kg·y
Surface water	1.1E-02	2.9E+01	3.4	36	ng/L
Waste	1.2E-02	31	3.7	39	mg/kg
WIP* bottom ash	7.0E-02	191	22	242	mg/kg
WIP* fly ash	1.0E-01	261	29	334	mg/kg
WWTP* effluent	4.9E-04	2.2	1.2E-01	2.4	µg/L
WWTP* sludge	3.1E-02	79	9.3E+00	100	mg/kg
			CH		
	Q15	**Mean**	**Median**	**Q85**	**Unit**
Air	2.7E-04	7.6E-01	9.4E-02	9.7E-01	ng/m3
Sediment	1.5E-01	450	50	560	µg/kg·y
N&U* soil	9.2E-05	2.5E-01	2.9E-02	3.3E-01	µg/kg·y
ST* soil	0	0	0	0	µg/kg·y
Surface water	2.0E-02	5.5E+01	6.7	69	ng/L
Waste	2.4E-02	69	8.5	87	mg/kg
WIP* bottom ash	4.0E-02	120	14	150	mg/kg
WIP* fly ash	5.0E-02	140	16	180	mg/kg
WWTP* effluent	6.6E-04	3.4	1.9E-01	3.6	µg/L
WWTP* sludge	3.7E-02	110	12	140	mg/kg

** N&U soil: Natural and urban soil; ST soil: Sludge treated soil; WIP: Waste incineration plant; WWTP: Wastewater treatment plant*

Hazard assessments in the surface water system were also conducted for these two ENMs using the PSSD approach. PSSD of nano-SiO_2 and nano iron oxides, respectively are displayed in Annex 1 and 2. The mean PNEC values were predicted to be 1028 and 218 µg/l for nano-SiO_2 and nano iron oxides, respectively. The corresponding mean values of the RCR in the EU for them are predicted to be several orders of magnitude smaller than 1 (0.0018 for nano-SiO_2 and 0.00012 for nano iron oxides). The low RCR can also be deduced from the comparison between the PEC and PNEC distributions (Figure 1). The distribution of RCR for Switzerland is shown in Figure A3. Therefore, the output of these two studies indicates that only a very limited risk is expected by for nano-SiO_2 and nano iron oxides at the current release level. Other ecological systems, e.g. soil, sediment, were not considered so far in the studies due to the lack of enough ecotoxicological data.

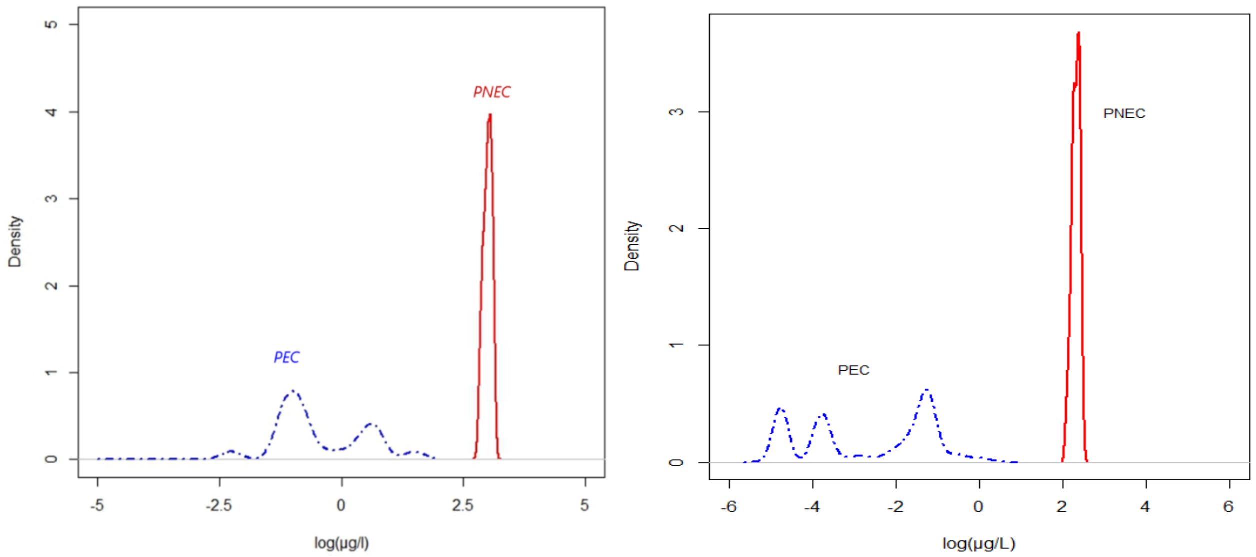
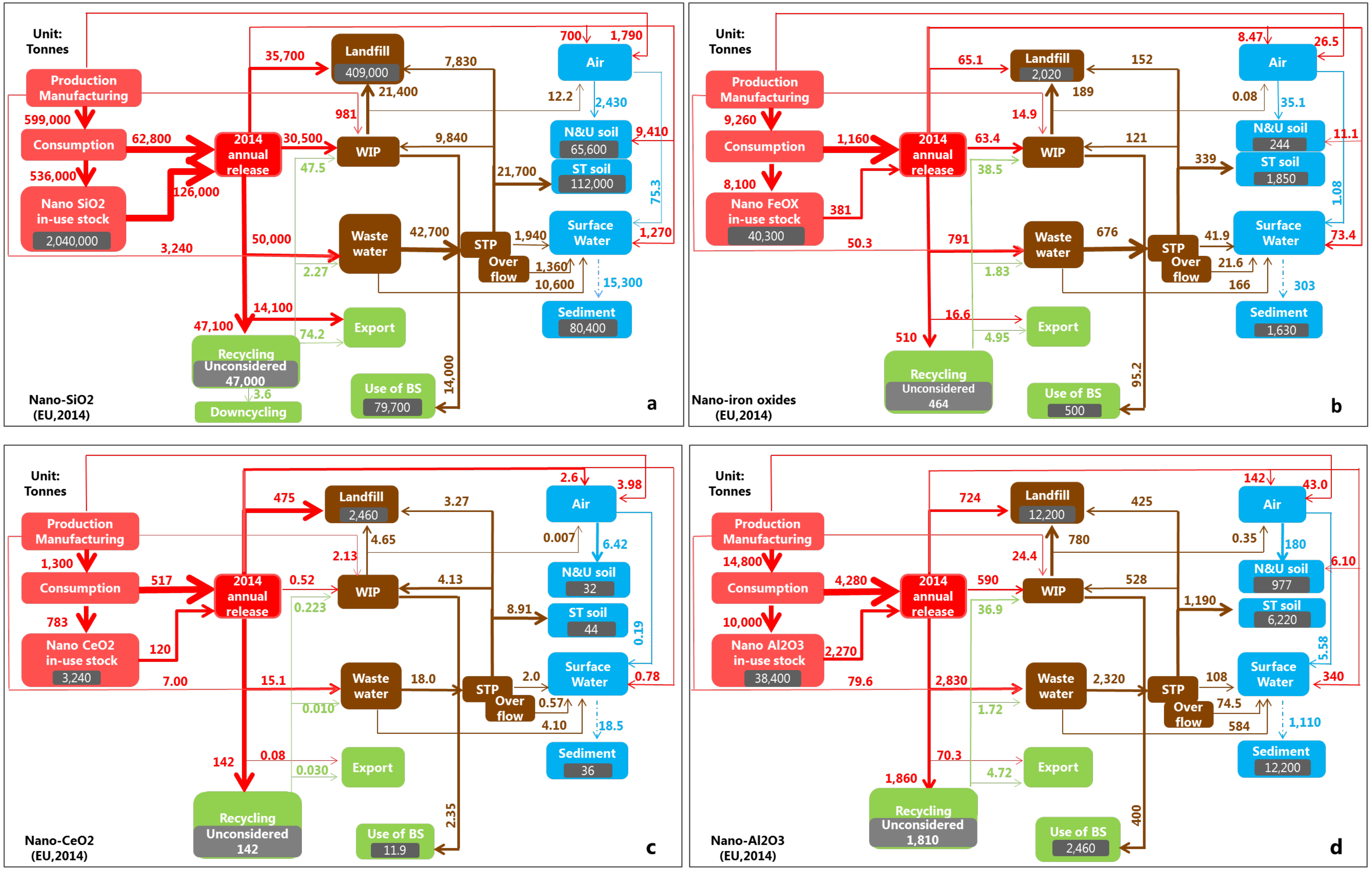


Figure 1. The predicted no-effect concentration (PNEC) distribution (in red) compared with the probability density distribution of predicted environmental concentrations (PEC) (in blue) for nano silica (left) and nano-FeOX (right) in the Swiss surface water system.

2.2 Dynamic modelling

The DPMFA model is more advanced compared to the static PMFA, because DPMFA is able to consider the change of the modelling system over time, incorporating a changing input over time and a dynamic release of ENMs from nanoproducts. The release and environmental concentrations were modelled using this model for nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots in the EU as a whole and six geographical sub-regions (Central Europe, Northern Europe, Southern Europe, Eastern Europe, South-eastern Europe, and Switzerland). The model showed that a large amount of ENMs are accumulated in stocks, e.g. soils and sediments. For example, 80,400 tonnes of nano-SiO_2 accumulated in sediments from 1990 to 2014 (see Figure 2).


Unit: Tonnes
Production Manufacturing
599,000
Consumption
62,800
536,000
Nano SiO2 in-use stock
2,040,000
126,000
2014 annual release
35,700
Landfill
409,000
21,400
7,830
981
30,500
WIP
9,840
47.5
3,240
50,000
Waste water
42,700
2.27
STP
1,940
Over flow
1,360
10,600
21,700
Air
700
1,790
12.2
2,430
N&U soil
65,600
9,410
ST soil
112,000
75.3
Surface Water
1,270
15,300
Sediment
80,400
14,100
47,100
Export
74.2
Recycling Unconsidered
47,000
3.6
Downcycling
Use of BS
79,700
14,000
Nano-SiO2 (EU,2014)
a
Unit: Tonnes
Production Manufacturing
9,260
Consumption
1,160
8,100
Nano FeOX in-use stock
40,300
381
2014 annual release
65.1
Landfill
2,020
189
152
14.9
63.4
WIP
121
38.5
50.3
791
Waste water
676
1.83
STP
41.9
Over flow
21.6
166
339
Air
8.47
26.5
0.08
35.1
N&U soil
244
11.1
ST soil
1,850
1.08
Surface Water
73.4
303
Sediment
1,630
16.6
510
Export
4.95
Recycling Unconsidered
464
Use of BS
500
95.2
Nano-iron oxides (EU,2014)
b
Unit: Tonnes
Production Manufacturing
1,300
Consumption
517
783
Nano CeO2 in-use stock
3,240
120
2014 annual release
475
Landfill
2,460
4.65
3.27
2.13
0.52
WIP
4.13
0.223
7.00
15.1
Waste water
18.0
0.010
STP
2.0
Over flow
0.57
4.10
8.91
Air
2.6
3.98
0.007
6.42
N&U soil
32
ST soil
44
0.19
Surface Water
0.78
18.5
Sediment
36
0.08
142
Export
0.030
Recycling Unconsidered
142
Use of BS
11.9
2.35
Nano-CeO2 (EU,2014)
c
Unit: Tonnes
Production Manufacturing
14,800
Consumption
4,280
10,000
Nano Al2O3 in-use stock
38,400
2,270
2014 annual release
724
Landfill
12,200
780
425
24.4
590
WIP
528
36.9
79.6
2,830
Waste water
2,320
1.72
STP
108
Over flow
74.5
584
1,190
Air
142
43.0
0.35
180
N&U soil
977
6.10
ST soil
6,220
5.58
Surface Water
340
1,110
Sediment
12,200
70.3
1,860
Export
4.72
Recycling Unconsidered
1,810
Use of BS
2,460
400
Nano-Al2O3 (EU,2014)
d

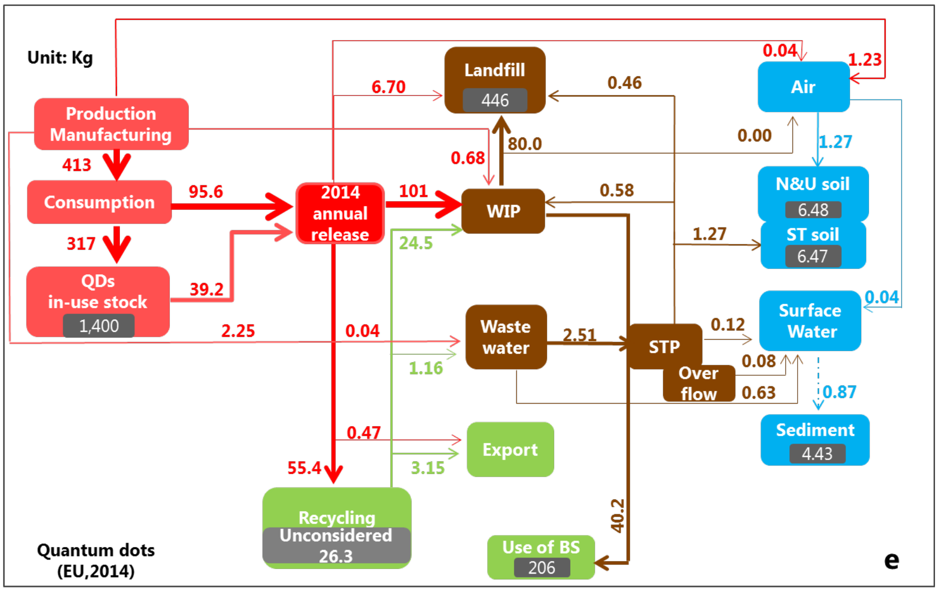


Figure 2. Mass flow diagrams for five ENMs in the EU in 2014 based on dynamic modeling: (a) nano-SiO_2, (b) nano-iron oxides, (c) nano-CeO_2, (d) nano-Al_2O_3, and (e) quantum dots. The red boxes represent the model input compartments. The green and brown boxes represent those waste treatment processes for which reuse or recycling processes are or are not involved, respectively. The blues boxes represent the environmental compartments. The dark grey boxes show the amount of ENMs held in stocks in each compartment up to 2014. A dashed line was used to represent the flow from surface water to sediment as two extreme scenarios were assumed: a) no transfer and b) complete sedimentation. All presented values are the means of the mass distributions and are rounded to three significant numbers. The unit for QDs is kg/y, while the unit for the other ENMs is tonnes/y. "Unconsidered" represents those products, of which the further behavior after recycling processes was not considered due to lack of data; WIP: waste incineration plant; STP: sewage treatment plant; BS: bottom ash; N&U soil: natural & urban soil. ST soil: sludge-treated soil.

Hazard assessments were performed for nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots using an PSSD method, as well as an environmental risk assessment based on the exposure level obtained from the DPMFA. The PNEC and the RCR values are presented in Table 3. The risks posed by these five ENMs were demonstrated to be nano-SiO_2 > nano-Al_2O_3 > nano iron oxides > nano-CeO_2 > quantum dots. However, all the RCR values for the considered ENMs are three to seven orders of magnitude lower than one.

Table 3. Mean values of predicted no effect concentrations (PNEC) and risk characteristic ratio (RCR)

ENM	PNEC (µg/l)	RCR (NE)	RCR (SEE)
nano-SiO_2	220 (196 - 248)	2.21E-03 (2.25E-05 - 7.16E-03)	1.13E-02 (9.85E-05 - 3.86E-02)
nano iron oxides	128 (84 - 169)	1.10E-04 (2.18E-06 - 6.27E-04)	3.64E-04 (8.25E-06 - 1.67E-03)
nano-Al_2O_3	120 (107 - 134)	3.21E-04 (6.68E-06 - 8.57E-04)	1.93E-03 (3.13E-05 - 6.41E-03)
nano-CeO_2	2.6 (2.1 - 3.2)	9.14E-05 (2.33E-06 - 4.09E-04)	3.6E-04 (7.88E-06 - 1.58E-03)
Quantum dots	0.32 (0.07 - 1.11)	2.43E-07 (1.75E-08 - 8.06E-07)	7.84E-07 (3.59E-08 - 3.13E-06)

Note: the values in the bracket are the 5th and 95th percentiles. NE: Northern Europe, SEE: Southeastern Europe

4. Conclusions

The output from this project provides an in-depth evaluation of the release of ENMs to technical and environmental systems and presents risk assessments towards organisms in surface water systems for nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots. Our risk assessment shows no overlap between the PEC distributions in fresh waters and PNEC distributions in all cases for these materials. Therefore, no risk is predicted under the estimated release level in 2020 for the five considered ENMs.

The results obtained in this project can serve as a starting point for further environmental fate modelling studies by providing regionalized data as input values. More importantly, the results from this work can also provide a scientific foundation to evaluate policies regarding environmental risks of ENMs.

Publication list

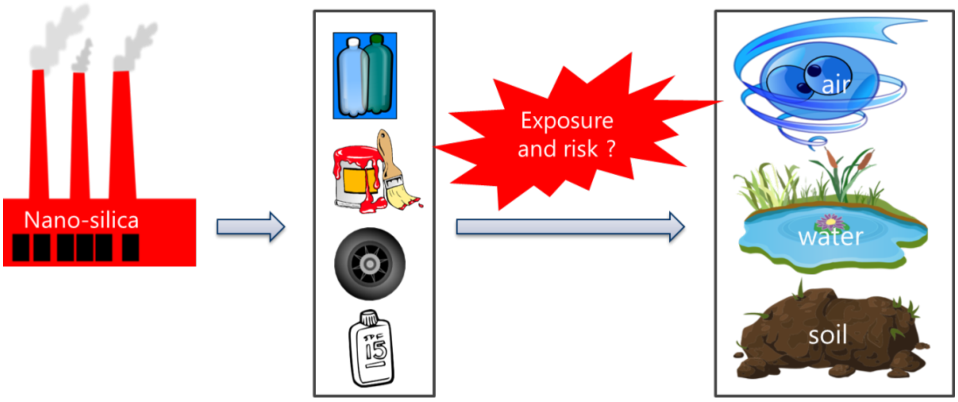
- **Wang Y**, Kalinina A, Sun T, Nowack B. 2016. Probabilistic modeling of the flows and environmental risks of nano-silica. *Science of the Total Environment* 545-546:67-76.

- **Wang Y**, Deng L, Caballero-Guzman A, Nowack B. 2016. Are engineered nano iron oxide particles safe? an environmental risk assessment by probabilistic exposure, effects and risk modeling. *Nanotoxicology* 10:1545-1554.

- **Wang Y**, Nowack B. 2017. Dynamic probabilistic material flow analysis of nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots in seven European regions. *Environmental Pollution*, under revision

- **Wang Y**, Nowack B. 2017. Environmental risk assessment for nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots. *Environmental Toxicology and Chemistry*, under revision.

Annex 1. Probabilistic modeling of the flows and environmental risks of nano-silica



Science of the Total Environment 2016, volume 545–546, page 67–76

Abstract

Nano-silica, the engineered nanomaterial with one of the largest production volumes, has a wide range of applications in consumer products and industry. This study aimed to quantify the exposure of nano-silica to the environment and to assess its risk to surface waters. Concentrations were calculated for four environmental (air, soil, surface water, sediments) and two technical compartments (wastewater, solid waste) for the EU and Switzerland using probabilistic material flow modeling. The corresponding median concentration in surface waters is predicted to be 0.12 μg/L in the EU (0.053-3.3 μg/L, 15/85% quantiles). The concentrations in sediments in the complete sedimentation scenario were found to be the largest among all environmental compartments, with a median annual increase of 0.43 mg/kg·y in the EU (0.19-12 mg/kg·y, 15/85% quantiles). Moreover, probabilistic species sensitivity distributions (PSSD) were computed and the risk of nano-silica in surface waters was quantified by comparing the predicted environmental concentration (PEC) with the predicted no-effect concentration (PNEC) distribution, which was derived from the cumulative PSSD. This assessment suggests that nano-silica currently poses no risk to aquatic organisms in surface waters. Further investigations are needed to assess the risk of nano-silica in other environmental compartments, which is currently not possible due to a lack of ecotoxicological data.

A1.1 Introduction

As a result of the development of nanotechnology, the production of engineered nanomaterials (ENMs) has increased significantly over the last decades and numerous applications penetrate almost every aspect of our life. Although nanomaterials benefit our everyday life, they can also have adverse impacts on humans and the environment [92]. Due to their small size and large specific surface area, nanoparticles are more likely than large-scale materials to penetrate cells, pass through biological barriers and generate free radicals and reactive oxygen species [93, 94]. Thus it is critical to evaluate human and environmental risks posed by ENMs, especially those with high production volume. This field has received a lot of attention from researchers and regulators [95].

Nano-silica represents silica particles and agglomerates with a size of the primary particles between 1 and 100 nm [96]. Synthetic amorphous silica (SAS) has been produced since the 1950s and has a worldwide production of several million tons [97]. Nano-silica is used in various pharmaceutical products, cosmetics, printer toners and food products because it can provide materials with a desired consistency and prevents separation of various ingredients [65]. In addition, it is also widely used in paints, surgical tools, medical equipment, textiles and all sorts of surfaces and substrates due to its high water repellence [66]. With desired surface and internal functionality, nano-silica also has a high potential for application in different biorecognition agents, e.g. antibodies, protein complexes, etc. [67]. Furthermore, the silica matrix has a negative charge and can provide a high number of electrostatic binding sites. This property is widely used for drug delivery purposes [98].

The production and use of nano-silica unavoidably cause environmental release of it. However, the knowledge about environmental exposure to nano-silica still remains scarce. Currently, only a few release studies are available that have investigated the release of nano-silica from products [99]. For example, the release of nano-silica from nanocomposites by three different release mechanisms has been studied [100]. Nguyen et al. (2012) have reported the release of nano-silica during the irradiation of polymer composites and Al-Kattan et al. (2015) and Zuin et al. (2014) have quantified the release of silica from paint containing nano-silica into water.

To date, there are no analytical methods available to detect trace concentrations of inorganic ENMs such as nano-silica in environmental samples [30]. Environmental exposure assessment therefore has to depend on modeling approaches to predict the environmental concentrations [104]. In the past few years, several models have been developed to assess release and environmental concentrations of ENM [34, 37, 38, 40, 41, 105]. Some modeling studies have also included nano-silica. Boxall et al. (2007) have modeled the environmental exposure to more than ten ENMs in the UK environment including nano-silica and the concentrations of nano-silica were predicted to be 0.0007 μg/l in water and 0.03 μg/kg in soil, assuming a 10% market penetration of the nano-products. Keller et al. (2013) have predicted a global release of 10,600 metric tons of nano-silica to soil and 2,100 metric tons to water in 2010. Furthermore, they have performed a local modeling study to calculate the release of nano-silica in effluents from sewage treatment plants (STP) in California (0.63-53 metric tons/year) and San Francisco bay (120-10,000 kg/year) in 2010 [40]. However, the methods used in these models did not consider the high uncertainties of almost all model parameters. A simplistic algorithm approach was applied in Boxall's model considering only one use and complete release. In Keller's model linear equations to calculate the concentrations of ENMs in the environment were used without considering the uncertainty of the input data. The probabilistic material flow model developed by Gottschalk et al. (2010) is able to consider the uncertainty of all model parameters. The release of a variety of different nanomaterials has been studied using such model [37, 41, 42].

The hazards of nano-silica have been investigated by several toxicological and ecotoxicological studies [65, 106-108]. Napierska et al. (2010) have evaluated both nano-silica and synthetic amorphous silica in *in vivo* and *in vitro* studies and apoptosis was only detected after exposure to nanoparticles (14 nm) and no effect was observed after treatment with micro particles (1-5 µm). Dekkers et al. (2013) described a higher toxic potential of nano-silica in food after intravenous injection than after oral administration and adverse effects were detected after exposure to 3 mg/kg body weight per day for intravenous injection in comparison with 1000-2000 mg/kg body weight per day/day for oral administration. The effects of nano-silica on aquatic organisms have also been investigated to some extent. Yang et al. (2014) for example have analyzed the ecotoxicological effect of nano-silica and bulk silica to *Daphnia magna.* The study indicated that nano-silica has a dose-dependent effect on *Daphnia magna* but bulk silica does not.

Risk assessments for nano-silica have been performed so far only for specific applications. One example is the risk assessment of a glass cleaner formulation (spray application) containing nano-silica [61]. In addition, Dekkers et al. (2013) have conducted a risk assessment of nano-silica in food. However, no environmental risk assessment covering a broad range of products containing nano-silica has been performed so far.

Considering the lack of exposure concentrations and the absence of generalized environmental risk assessments for nano-silica, the aim of this study is to evaluate the environmental release and risks of nano-silica using probabilistic material flow and environmental risk assessment models.

A1.2 Methods

A1.2.1 General layout of the model

Material flow analysis (MFA) was used to determine the flows and stocks of nano-silica. The principle of MFA is to express all available information concerning a material presenting in processes, flows and stocks. Figure A1.1 presents the general structure of the MFA model which is based on Gottschalk et al. (2009) and Sun et al. (2014). The idea behind the model is to trace nano-silica during all phases of the product life cycle (e.g. production, manufacturing, use, and disposal) and to quantify the transport to technical compartments and transfer from technical compartments to the environment. The system boundaries considered in this work are the EU and Switzerland.

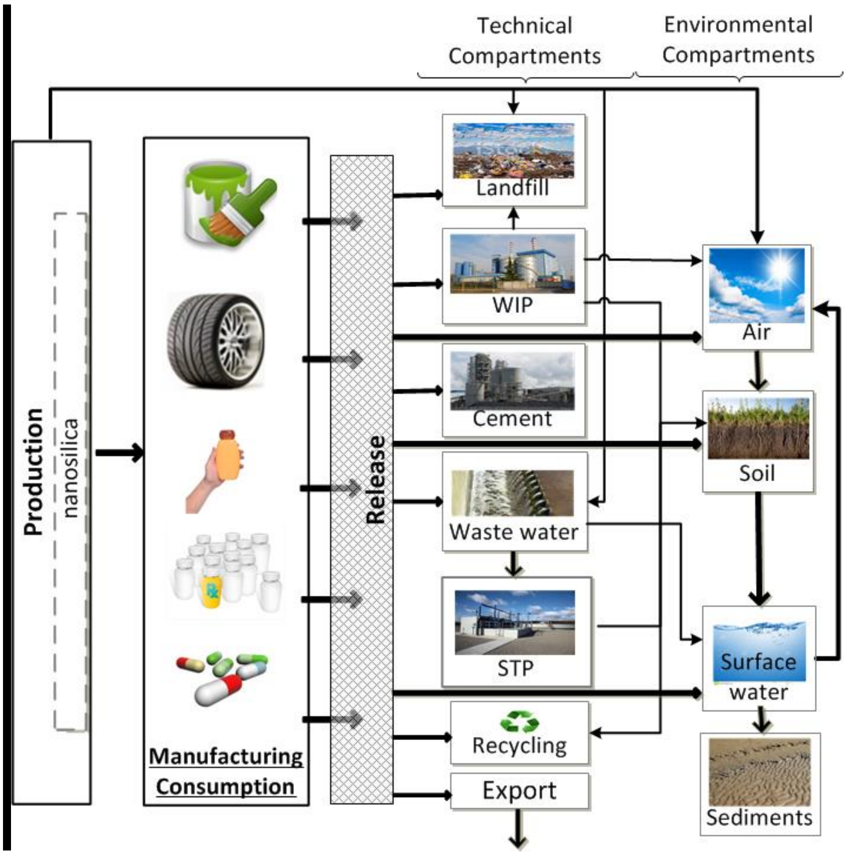


Figure A1.1. Framework of the material-flow model. The principle is to trace the flows of nano-silica from production, manufacturing, and use to disposal and to quantify the transport from technical compartments (landfill, waste incineration plant (WIP), cement, sewage treatment plant (STP), recycling and export) to environmental compartments (air, soil, surface water and sediments).

The model contains four environmental compartments: soil (split into general soils and sludge treated soils); surface water; air; sediments; and seven technical compartments: production, manufacturing, consumption (PMC); sewage treatment plants (STP); waste incineration plants (WIP); landfills; recycling; cement; export.

A1.2.2 Model input parameters

A1.2.2.1 Input data treatment

Due to limited knowledge about input parameters (production, allocation of nano-silica to product categories and transfer coefficients), the output of the model is inherently uncertain. To deal with this high uncertainty, all input values were introduced into the model as probability distributions. Two types of distributions – triangular and uniform distributions - were used to process the input parameters. The principle of applying these distributions to different data sources was shown in Sun et al. (2014).

The Degree of Belief (DoB) parameter was assigned to production volumes and STP removal efficiency from different sources as a measure of their reliability. Two levels of DoB were used: 20% and 80% DoB, which were assigned to the sources of low and high reliability, respectively. This method also was used in a previous study [41]. High reliability was assigned to data from reviewed studies with explicit focus on the parameters of interest or with detailed explanation for data collection. Data obtained from personal communications and market reports were also attributed to having high reliability. Other information sources such as presentations or booklets, where only general information was given without detailed description concerning the used methods, were considered as low reliability sources, as well as studies published before 2007. In the probabilistic model, a DoB of a parameter was reflected by a larger or smaller sampling size for the corresponding parameter, which is in proportion to the assigned DoB when applying the Monte Carlo (MC) approach [109]. For instance, if a data point had a DoB of 20%, then 20,000 samples out of the total 100,000 samples were assigned to this data point. The underlying methodology is described in Gottschalk et al (2010) and Sun et al (2014).

To perform the exposure assessment the predicted environmental concentrations (PECs) have to be quantified. PECs were calculated by dividing values of the modeled material flows of nano-silica to the environmental compartments by the volume or mass of the respective compartment. The volumes of environmental and technical compartments for the EU and Switzerland were taken from Sun et al (2014). The obtained concentrations are average values for each compartment assuming that the environment is a homogeneous and well-mixed system. For the air and water compartment, a retention time of 10 days and 40 days was considered respectively [110, 111]. The concentration in surface waters was calculated assuming no sedimentation, while for sediments 100% sedimentation was considered. The concentration results for these two compartments therefore constitute mutually exclusive worst-case scenarios.

A1.2.2.2 Nano-silica production volume

The production volume is the primary input parameter for the material-flow model. We have considered ten studies, which have reported the production volume of nano-silica. Global and regional (e.g. France) production volumes were extrapolated to the EU and Switzerland in proportion to PPP GDP (Gross Domestic Product per capita based on Purchasing Power Parity). PPP GDP, which is the Gross Domestic Product (GDP) converted to international dollars using purchasing power parity rates, was used as scaling factor to extrapolate production volumes from one country or region to another. Compared to the GDP, which was employed as scaling factor in previous studies [41], the PPP GDP takes into account the relative cost of local goods, services and inflation rates, which could be a better measure of the purchasing power of a country. A similar approach applying the Inequality-adjusted Human Development Index has also been used by other modeling studies [39]. As a result of using such factors, extrapolated production volumes of the EU and Switzerland can better present the actual values. A DoB was assigned to each data source as shown in the Table A1.1.

A1.2.2.3 Product distribution

The properties of a product incorporating nano-silica together with the shares of nano-silica in different product categories determine where and how much of the ENMs are released. All data from available product inventories were combined to estimate the allocation of nano-silica to different product categories. The structured methodology to record all relevant information about products (weight, the amount of nano-silica contained and number of products) was applied as described in [35, 37]. For each product category, maximal, minimal and mean values of the nano-silica share were defined. These values formed a triangular distribution of the share of nano-silica in each product category. This category is called 'own categories' with all details shown in Table A1.3 in the appendix 1. The product categories were defined based on categories in the French registration report (ANSES, 2013) together with data from the databases and inventories described. The following databases were used as data sources: the inventory of the Woodrow Wilson Centre for Scholars' Project on Emerging Nanotechnologies [112]; the ANEC/BEUC Inventory [113]; the BUND inventory [114]; and the Nanodatabase [115]. In addition, patent Information about products containing nano-silica was collected from two patent databases: the American US Patent & Trademark Office [116] and the European Patent Register [117]. EC21 [118] was also used to count the number of products containing nano-silica for different utilization fields on the market.

In addition, it is also important to consider product categories for nano-silica from previous studies to estimate the allocation of nano-silica to different product categories. In this work we also used the categorization from Keller et al. (2013), in which six categories are listed (automotive, catalysts, electronics & optics, energy & environment, sensors and coatings, paints & pigments) and they have equal contributions to the global production of nano-silica. We assumed that nano-silica is allocated to the categories in the EU in the same way as worldwide. However, it is necessary to point out that the categories developed in the Keller et al. (2013) study are industrial sectors rather than product categories. For instance, the category "energy & environment" has 20% of the total nano-silica production and it is not clear which products it actually contains. This categorization method is named "Keller's categories" in this study.

A1.2.2.4 Release and transfer factors

The transfer coefficients describe what share of nano-silica in a product is released to different technical or environmental compartments. Transfer coefficients between and within compartments were estimated based on the current knowledge of behavior and fate of nano-silica in these compartments [96, 101]. If there was no release information regarding those product categories containing nano-silica, we adopted transfer factors from release studies for products with similar properties or for the same product categories but containing metal oxide ENMs with similar behavior, e.g. textile and paint [119-121]. For Keller's categories, the transfer coefficients were reassessed based on those determined by Sun et al. (2014) instead of taking the transfer factors from Keller et al (2013). This is because in their evaluation over 80% of nano-silica was estimated to end up in landfills, but in the EU and Switzerland many products containing nano-silica are recycled. The transfer coefficients for "own categories" and "Keller's categories" are given in Table A1.4 which also contains a complete description how the transfer coefficients were obtained.

The removal efficiency of STP and WIP was modeled based on data from the literature. Table A1.5 lists the removal efficiency of nano-silica during wastewater treatment from different studies that were combined into a probability distribution for removal efficiency (Figure S1.1). The removal efficiency of nano-silica during waste incineration was assumed to be equivalent to CeO_2 [122].

A1.2.3 Hazard assessment

The goal of the hazard assessment is to identify the hazards of nano-silica towards the ecosystem. Hazards can be identified by the predicted no effect concentration (PNEC) which is the threshold of not having adverse effects on ecosystems. Based on the European regulation [48], species sensitivity distributions were used for deriving the PNEC. Probabilistic species sensitivity distributions (PSSD) [54, 56] were computed by combining the probabilistic distributions of all single species into oneprobability density function. The value of the 5th percentile of the cumulative PSSD (Hazard effect 5, HC5) was then used as the PNEC value for further analysis according to[48]. In our probabilistic approach not a single PNEC value but a distribution of the PNEC values was derived from the PSSD by abstracting values from 10,000 runs. The detailed methodology to derive the PSSD and PNEC distribution is described in Coll et al. (2015).

According to the ECHA guidance[48], it is advisable to collect a dataset of at least 10 (preferably more than 15) concentration endpoints for different species from at least 8 taxonomic groups to develop a SSD. We were able to collect 20 concentrations from 8 species for 4 taxonomic groups. Table 2-1 shows that twenty endpoints (EC20, NOEC, IC50, EC50, LOEC, HONEC and LC50) from eight different species were found for nano-silica. The detailed dataset is presented in the Table A1.7 with the types of endpoint, names of the tested organisms and their taxonomic groups, and exposure time. NOEC values were obtained by transforming each endpoint concentration by two assessment factors (AF)[48]. One factor is used to extrapolate observed effects into no-observed effect concentrations (NOEC). An AF of 10 was used for LC/EC25-50, an AF of 2 for LC/EC10-20, and an AF of 1 for LOEC, LED, MIC, HONEC and NOEC values. The second AF of 10 was considered to convert the short term to long term effects based on established thresholds for test duration [48].

Table A1.1. Taxonomic groups and species, which were used to compute endpoint concentrations for nano-silica

Taxonomic group	**Test organism**	**Endpoints available**
Algae	*Scenedesmus obliquus*; *Pseudokirchneriella subcapitata, Chlorella kessleri*	12
Invertebrates	*Daphnia magna*	3
Unicellular	*Saccharomyces cerevisiae*	1
Vertebrates	*Carassius auratus; Poecilia reticulata; Danio rerio*	4

A1.2.4 Risk characterization

According to the ECHA guidance, environmental risk is quantified by comparing the exposure levels to the hazard effects. The risk is characterized by the risk characterization ratio (RCR), which is obtained by comparing the exposure concentrations with computed PNEC values. The RCR was calculated using the following equation 1 [60]:

$$RCR= PEC/PNEC \qquad \text{Eq. (1)}$$

An RCR is lower than 1 indicates that the risk is controlled for the particular environmental compartment, route and conditions. An RCR that is higher than 1 shows the needs to implement additional risk management measures, or ban production or usage of this material [60]. Although the methodology was designed for conventional chemical substances or mixtures, the same framework can be applied to nanomaterials [123].

Equation 1 is used for a single value of PEC and a single value of PNEC in a deterministic approach. The same calculation principle was applied in the probabilistic approach employed in this study. The RCR distribution for a particular compartment was obtained by dividing all values of the PEC distribution of a compartment by all values of the PNEC distribution of the same compartment [124].

A1.3 Results

A1.3.1 Exposure assessment

A1.3.1.1 Production volume

All available sources with the information of production volumes for nano-silica were analyzed to estimate the production volume in the EU and Switzerland. The sources and scaling factors from global to regional production volume are presented in Table S1.1. The probability distribution of the production volume was computed out of all collected data by applying the MC method [36]. Figure A1.2 shows the probability distribution of the production volume in the EU in 2014 with mean, mode, median, quantile 0.15 and quantile 0.85 (Q15 and Q85) marked. The median value of production of nano-silica in the EU is 5,800 tons per year (the corresponding distribution and values for Switzerland are given in Figure S1.2.) Due to the very large range of reported production volumes, the probabilistic distribution curve has a long tail. In the model the complete distribution was used, considering also the likely range reflected by the 15-85% interval. The median value is used as the representative of an entire distribution, because it is less affected by the skewed raw data.

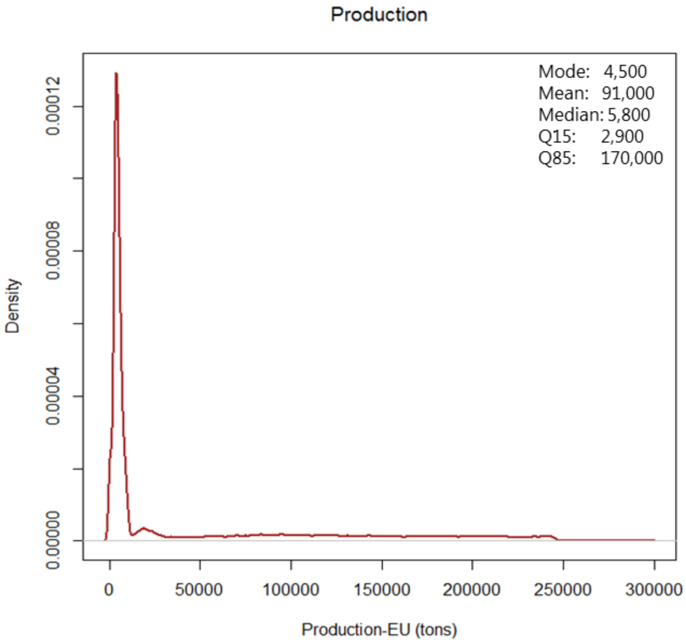


Figure A1.2. Probability distribution of the nano-silica production in the EU in 2014 in tons/y, all values are rounded to two significant numbers.

A1.3.1.2 Allocation of nano-silica to product categories

Some product categories have almost negligible percentage of nano-silica in the total production, only nine categories from the “own” categorization are shown in Figure A1.3. Other categories with less than 0.5% of the total nano-silica production were included into the 10th category as “Others”. The largest fraction in the category “Others” belongs to the category “Adhesives and sealants”. The minimal, mean and maximal shares of nano-silica in different products in the “own” categorization which were taken from different inventories and sources as well as shares for the Keller’s categories are shown in Table S1.2. The category “Paints” was found to be the largest product category containing nano-silica followed by the categories “Polymer” and “Coating”. Only two consumer products containing nano-silica were found in the Nanodatabase, which is the most up-to-date database of engineered nanomaterials used in consumer products. Those two are allocated to the categories “Paints” and “Adhesives, sealants” categories. A Wide application of nano-silica in coatings is reflected in a high share of the category “Coating”. The results correlate well with literature information about the applications of nano-silica in different products, and the characteristics (properties) that nano-silica gives to products [65, 125, 126].

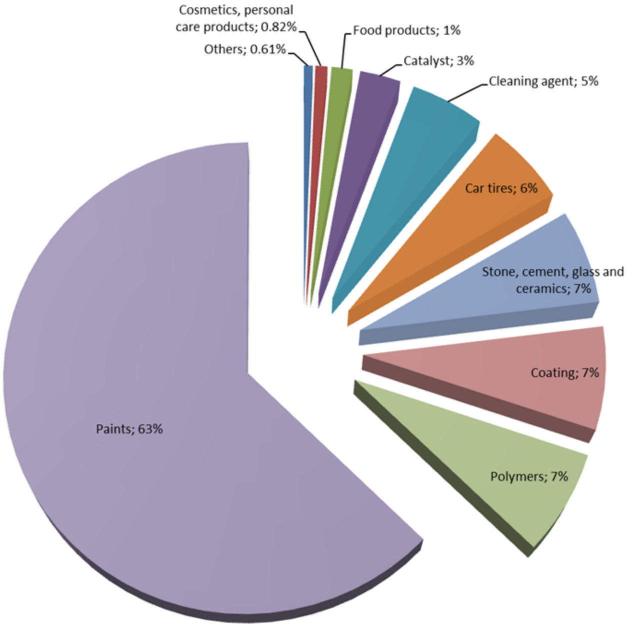


Figure A1.3. The largest nine product categories containing nano-silica with their shares of the total production volume

A1.3.1.3 Nano-silica flows

Based on the production volume and the product distribution together with the transfer factors given in Table S1.4, we calculated the nano-silica flows for the EU and Switzerland (Figure A2.4). The mean value of each flow is given in metric tons/year in the flow chart. In these charts mean values are presented because the mass balance is not obtained with median values. In Table A1.6 the mode, mean, median, Q15 and Q85 values are also given to allow a better evaluation of the spread of the values. The model has one total input flow, which is the annual production of nano-silica. Export from the geographic region represents the only output of the modeled system. The amount of nano-silica exported from the EU is almost negligible and contributes to less than 3% of the annual production (the value is 3.6% for Switzerland).

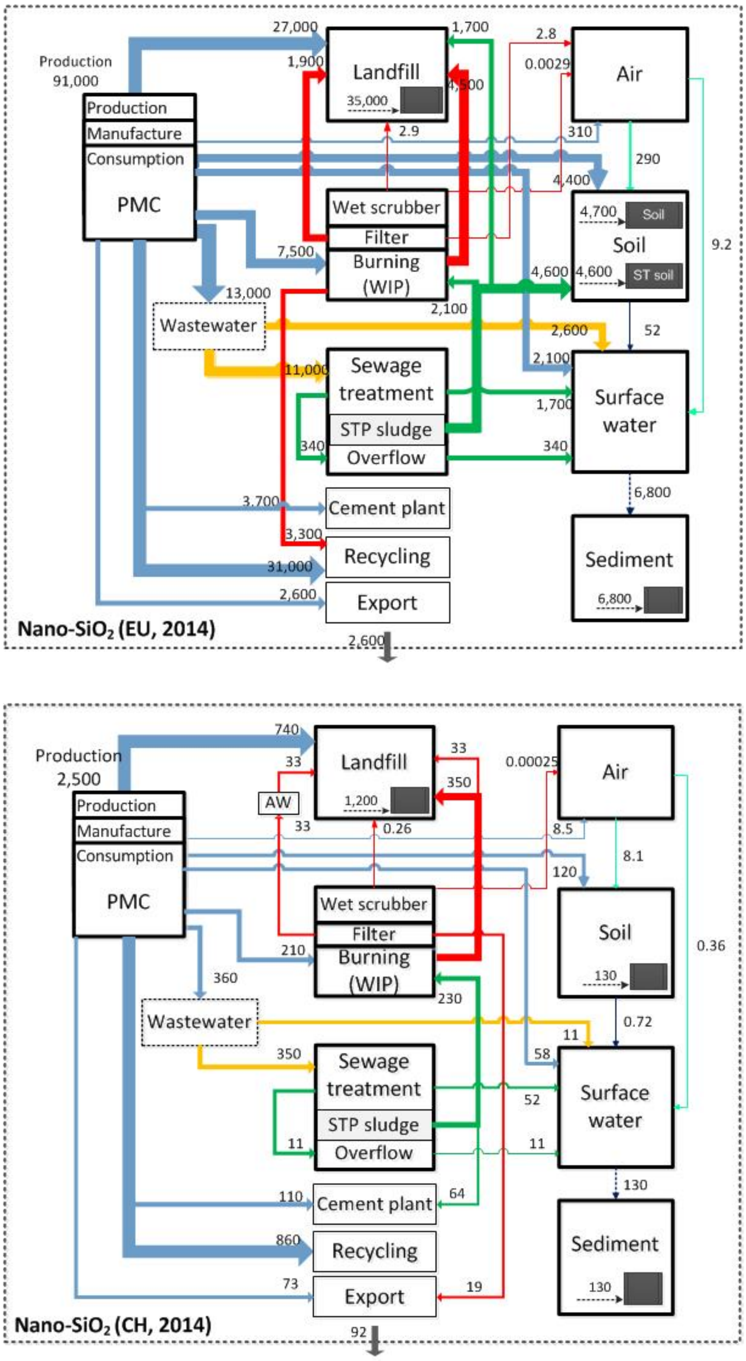


Figure A1.4. Nano-silica flows in the EU (top) and Switzerland (bottom). The mass of inflow and outflow of the model are balanced in terms of the mean values of the probability distribution (tons/year), which are noted on the flow chart and the flows given in the same color are the outflows from the same compartment. All values are rounded to two significant numbers; therefore the input and output flows from one compartment might not be balanced.

Shares of emissions to different compartments are strongly correlated with the specific aspects of the products application, as well as with the life cycles of the products. For instance, the release from paints and polymers mainly goes to landfill and recycling and around 70% of nano-silica is applied in paints and polymers. Therefore, the two dominant flows from Production, manufacturing and consumption (PMC) are to landfill (27,000 tons/a) and recycling (31,000 tons/a) in the EU. Together with the flow to wastewater, they account for 77% of the nano-silica mass which released from PMC. The "Landfill" and "Recycling" compartments are technical compartments where the nano-silica mainly goes. However, leachates from landfills and behavior of nano-silica during recycling were not considered due to lack of sufficient data in our model.

Nanomaterial flows from wastewater to STP are mostly captured in the sludge and are eventually transported to WIP, landfill and soil when part of sludge is used as fertilizer (in the EU but not in Switzerland). The main flows of nano-silica to surface water are from PMC (2,600 tons/year) and wastewater (2,100 tons/year) that is not connected to STP. Another considerable release to surface water is through sewage treatment plants (1,700 tons).

Soil, landfill and sediment are the compartments where nano-silica tends to accumulate. The air is the least affected compartment by nano-silica. The main material flow to air is from PMC (310 tons/year), and other trivial flows are from WIP both in the EU and Switzerland.

A1.3.1.4 Predicted concentrations in technical and environmental compartments

Probabilistic distributions of concentrations for the environmental compartments were determined for air, surface water, sediment, soil and sludge treated soil. In addition, nano-silica concentrations were also calculated for STP sludge, treated effluent from STP and municipal solid waste and the residues from waste incineration (bottom ash and fly ash). The predicted mean, median and mode values, as well as 15th and 85th percentiles of each distribution for the EU and Switzerland are presented in Table A1.2. The units of concentrations in soil, sludge treated soil and sediment correspond to the yearly increase. Under Swiss regulation, sewage sludge is transported to cement plants or WIP, instead of going to agriculture. Therefore, there is no sludge treated soil in Switzerland.

Among the environmental compartments, sediments have the highest yearly accumulation both in the EU (0.43 mg/kg.y) and Switzerland (0.60 mg/kg.y), followed by sludge treated soil (420 µg/kg.y) in the EU and soil (0.43 µg/kg.y in the EU and 1.3 µg/kg.y in Switzerland). The lowest concentration was found in air, which can be explained by the very low release of nano-silica to the air compartment and short retention time. It is necessary to point out that all concentrations in this model only represent the amount of nano-silica in the compartments in 2014 without considering the accumulation of nano-silica in the environment.

Table A1.2. Mean, mode, median values and 15th and 85th percentiles of predicted nano-silica concentrations (or yearly increase in concentration) in different environmental and technical compartments

			EU			
Compartment	**mean**	**mode**	**median**	**Q15**	**Q85**	**Unit**
Soil	6.2	0.42	0.43	0.14	11	µg/kg.y
Sludge treated soil	6,300	370	420	160	11,000	µg/kg.y
Surface water	1.9	0.037	0.12	0.053	3.3	µg/L
Sediment	6.7	0.13	0.43	0.19	12	mg/kg.y
Air	1.9	0.07	0.12	0.0029	3.5	ng/m^3
STP Effluent	56	1.3	3.8	0.61	83	µg/L
STP sludge	940	36	63	25	1,600	mg/kg
Solid waste	160	2.9	10	4.4	280	mg/kg
WIP bottom ash	1,000	12	68	28	1,800	mg/kg
WIP fly ash	1,400	61	93	37	2,400	mg/kg
			CH			
Compartment	**mean**	**mode**	**median**	**Q15**	**Q85**	**Unit**
Soil	21	0.64	1.3	0.42	34	µg/kg.y
Sludge treated soil	--	--	--	--	--	µg/kg.y
Surface water	2.8	0.19	0.17	0.064	4.4	µg/L
Sediment	9.8	0.66	0.60	0.23	16	mg/kg.y
Air	5.6	0.21	0.3	0.00015	9.5	ng/m^3
STP Effluent	900	43	57	8.7	1,200	µg/L
STP sludge	1,400	83	86	35	2,300	mg/kg
Solid waste	75	1.6	4.5	1.9	120	mg/kg
WIP bottom ash	920	23	54	24	1,500	mg/kg
WIP fly ash	1,000	77	63	26	1,700	mg/kg

A1.3.2 Hazard characterization

Twenty ecotoxicological endpoint concentrations were collected from eight different species. The species sensitivity spanned several orders of magnitude. It is found that *Chlorella kessleri* (Algae) has the lowest sensitivity to nano-silica [107], whereas *Pseudokirchneriella subcapitata* (algae) has the highest sensitivity [127]. Exposure time also was shown to be a crucial parameter for aquatic toxicity of nano-silica [128]. The size-dependent toxicity of nano-silica to *Chlorella kessleri* was investigated and it was found that the smaller particles are more toxic, which was reflected by the half maximal inhibitory concentrations (IC50) [127]. The IC50 for particles of 5 nm was 8,000 mg/l and it increased to 91,000 mg/l as the size went up to 78 nm.

The PSSD was computed only for the surface compartment, since endpoint concentrations were available only for aquatic organisms. The distribution was built by combining sensitivity distributions for eight individual species. The modeled cumulated PSSD is shown in Figure A1.5 as well as the measured data points and geometric mean values for the different species.

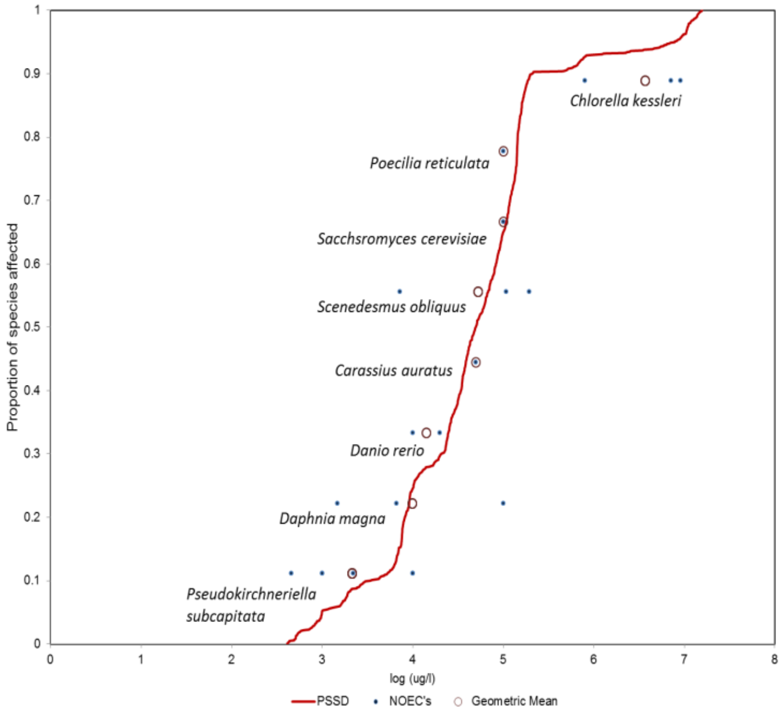


Figure A1.5. Cumulative probabilistic species sensitivity distribution (PSSD) of nano-silica in surface water based on NOEC values. The small blue points represent the single values of the endpoint concentrations, the empty circles are the geometric mean values (for illustration only, not used in the modeling) and red curve is the PSSD.

The PNEC values were obtained by extracting the 5th percentile from the PSSD. The computational procedure for the cumulative form of PSSD was repeated 10,000 times and 10,000 PNEC values were therefore obtained. All these values were used to derive the probability density distribution of PNEC. The median PNEC value is calculated to be 1,023 µg/L (mode value is 1,030 µg/L, mean value is 1,028 µg/L, Q15 is 787 µg/L, Q85 is 1,265 is µg/l).

A1.3.3 Risk calculation

The comparison between the PEC and the PNEC distributions is shown in Figure A1.6 for the EU (the distribution for Switzerland is shown in Figure A1.3). No overlap was observed in both cases. The RCR values were calculated by dividing all values of the PEC distribution by all values of the PNEC distribution. Table A1.3 gives the mean, mode, median, Q15 and Q85 values of RCR for the EU and

Switzerland. The calculated median RCR values are 0.00012 for the EU and 0.00017 for Switzerland. Figure S1.4 shows the distribution of the RCR for the EU and Switzerland. It is obvious that all RCR values are much lower than 1 within both system boundaries.

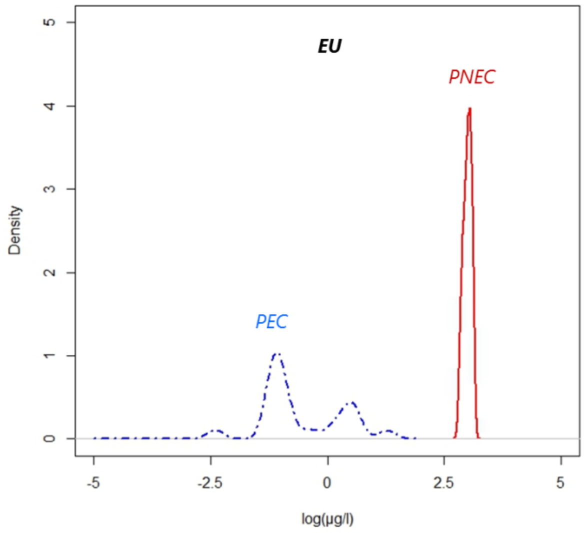


Figure A1.6. The Predicted No-Effect Concentration (PNEC) distribution in red compared with the probability density distribution of Predicted Environmental Concentrations (PEC) in blue for nano-silica in the EU surface waters.

Table A1.3. Risk Characterization Ratio of nano-silica in the surface water compartment in the EU and Switzerland

	mean	mode	median	Q15	Q85
RCR (EU)	0.0018	$5.8 \text{ X } 10^{-5}$	0.00012	$5.1 \text{ X } 10^{5}$	0.0034
RCR (Switzerland)	0.0031	$5.4 \text{ X } 10^{-5}$	0.00017	$6.9 \text{ X } 10^{-5}$	0.0046

A2.4 Discussion

This work predicted concentrations of nano-silica in several environmental and technical compartments by using probabilistic material flow modeling. In addition, the possible environmental risk of nano-silica was quantified in the surface water compartment by using probabilistic risk assessment. The study did not focus on one specific application but rather considered all possible product categories incorporating nano-silica. Hence the results here provide overall knowledge of environmental risk posed by nano-silica.

The estimation of the production volume of nano-silica was based on ten different sources, which provide values spanning a large range. The highest value was from the French registration report (scaled to the EU: 836,467 tons/year) [129], which is almost ten times more than the global production volume used in the studies by [40, 130]. This large spread of data is likely caused by different considerations of categorizing nano-silica and bulk silica. Silica incorporated in a “nanostructured material” can be either counted as nano-silica or as conventional material in different inventories for production estimates [131]. It can be expected that current and future inventories are based on the same EU definition for nanomaterials and thus this issue of definition could be resolved in the near future by using a unified description.

The flows to and concentrations of ENMs in the environment are determined by the properties of the products and the contained ENM. Our modeling results show the released nano-silica is distributed mainly to landfills and recycling. Keller et al. (2012) predicted that more than 86% of nano-silica ends up in landfills. The differences of the release behavior are because the product categories defined and estimated transfer factors are not the same between the current study and the previous work. The categories used by Keller et al (2012) are rather technological sectors than product categories. Thus, the estimation of transfer factors of those technological sectors is more difficult because they contain products with very different life cycles. Another explanation for the difference is that different ways of waste handling were considered: recycling, not mentioned in Keller’s model, is a non-negligible compartment where about 38% of nano-silica ends up in this study. However, we simplified the model by not considering the fate of nano-silica in the recycling system. Caballero-Guzman et al. (2015) provided information on the further fate during recycling by modeling the flows of nano-TiO_2, nano-ZnO, nano-Ag and CNT in the recycling system in Switzerland. The results showed that the largest portion of the nano-flows in recycling processes for the considered materials is subsequently properly handled in incineration plants or landfill as wastes. This is because ENMs are usually incorporated in material fractions that are not targeted in recycling processes but are considered as waste. A small fraction of ENMs is eliminated or kept in materials that are exported to further recovery processes and go back to productive processes of the economy in a limited number of sectors. Hence, our material flow model could be improved by including the behavior of nano-silica during recycling phases. However, currently no data are available for the EU to parameterize the system.

The modeled PEC of nano-silica in surface water is 0.12 µg/l (EU) with a likely range from 0.053-3.3 µg/l (15-85% quantile). These values can be compared to the previous estimates that were based only on simplistic assumptions. Michel et al. (2013) estimated local concentration of 1 µg/l of nano-silica in surface water. This estimation was based on the conservative assumption that 10% of the total consumption (18,000 tons /year) of nano-silica is released to the environment. Boxall et al. (2007) predicted the concentration of nano-silica to be 0.0007 µg/l in surface water in the UK. They assumed that nano-silica is only used in paints with 10% market penetration. The concentration of nano-silica in surface water in the Los Angeles region was modeled to be 0.03 µg/l and Keller and Lazareva (2014) predicted nano-silica concentrations in the San Francisco Bay for STP effluents in the range of 0.1-9 µg/l. In general, the differences in the concentrations correspond to differences in the estimated production volumes as well as the transfer factors to the water system. In the studies of Liu and Cohen (2014) and Keller and Lazareva (2014), nano-silica is more likely to flow to STP from paints coatings & pigments (60% to STP), automotive (90%) and catalysts (90%), and this in fact can result in a higher concentration in the surface water compartment. A large fraction of nano-silica is transferred to soils from electronics & optics (90%), environment & energy (90%) and sensors (95%) in the study of Liu & Cohen, 2014. This probably can explain why the PEC in soil (around 60 μg/kg) is higher than that of the current study (0.43 μg/kg·y in the EU and 1.3 μg/kg·y in Switzerland). The study by Boxall et al. (2007) predicted a concentration of nano-silica in soil of 0.03 μg/kg in the UK. Keller and Lazareva (2014) also predicted the nano-silica flow to the European environment, with a range of 282 to 1,753 tons released to soil. Overall the concentration is decided by production volume and transfer factors to different compartments.

The nano-silica concentrations predicted in our study for surface water and sediment are both for worst-case scenarios. We did not model any environmental fate processes but assumed for the surface water concentration no sedimentation and for the sediment concentration complete sedimentation. Only a fully mechanistic environmental fate modeling including heteroagglomeration, sedimentation and sediment transport will enable to predict nano-silica concentrations in water and sediments more precisely. The available fate models [130, 133, 134] rely strongly on input data to the environmental compartments that are provided by the material flow modeling presented in this study.

The environmental and technical concentrations predicted in this work correspond to a regional assessment and are based on well-mixed compartments and follow as such the ECHA guidance [135]. A next step in the exposure assessment would be to regionalize the emissions which would also allow us to identify hotspots [136, 137]. The flows presented in this work will provide the basic data needed to regionalize flows.

In the risk assessment conducted by Michel et al. (2013) for nano-silica in a glass cleaner formulation, the PNEC value was predicted to be 100 μg/l, which is ten times less than the value obtained in this paper (1,033 μg/L). This is due to the different computational process: more data was available for the construction of the PSSD while in the paper by Michel the PNEC value was obtained by applying a safety factor of 1,000 to the lowest toxicity value (algae, EC_{50} of 100 mg/l). In the PSSD approach smaller safety factors are used due to higher data availability.

The PEC values of nano-silica in the surface water compartment do not overlap with the PNEC values. The PEC values in the possible range of 0.053-3.3 μg/L in the EU and 0.06-4.46 μg/l in Switzerland (15-85% quantile) are three orders of magnitude lower than the PNEC values that are in the range of 787-1,265 μg/L (15-85% quantile). Concentrations of nano-silica are therefore not critical within both system boundaries. Moreover, the median values for the risk characterization ratio were estimated to be 0.00012 in the EU and 0.00017 in Switzerland. Hence, harmful effects caused by the exposure to nano-silica in the surface water compartment are negligible in the current situation.

Meanwhile, ecotoxicological studies and environmental risk assessments for bulk silica have also been performed. It has been reported that bulk silica is nontoxic to *Pseudokirchneriella subcapitata* up to 1 g/L [138]. Yang et al. (2015) have compared the effects of nano and bulk silica on *Daphnia magna.* It was found that at exposure concentrations of 400 mg/L, the rate of death was 43% for nano-SiO_2 exposure group while for bulk SiO_2 the i mortality rates were 6%.. The risk assessment report from the Synthetic Amorphous Silica and Silicate Industry Association has shown that bulk silica and silica gel are non-active substances in the environment and they are practically non-toxic to non-target organism [97]. Although nano-silica seems to be more toxic that bulk silica, it poses no environmental risks at the current production amounts and use pattern.

It is necessary to point out that in our risk assessment model uncertainties exist in the exposure assessment as well as in the hazard assessment. The applied probabilistic approach helps to deal with uncertainty of the input data (production volumes and transfer coefficients). However, other sources of uncertainty or variability might still remain, for example, the dynamics of the system or fate processes. If processes such as dissolution were considered, the emission routes of nano-silica could be different. Al-Kattan et al. (2015) for example, reported that silica released from paint fragments was mainly in the dissolved form, indicating significant dissolution during the experiment. Data on the characterization of the released materials and the quantification of the dissolution process is therefore needed because the dissolution can substantially lower the released concentration of nano- silica. However, this would further decrease the PEC values and thus further increase the gap between PEC and PNEC.

High uncertainty of effect concentrations for the tested aquatic organisms was found during the literature review. Lack of essential information about testing conditions is missing in some reports. This might introduce uncertainty in the results, as it is possible that the same organism shows different effects under different environmental conditions such as pH, temperature, extent of UV radiation, etc. The uncertainty of ecotoxicological effect concentrations were handled by computing probability distributions for each tested organism and considering all available endpoints. However, a better understanding of the effects of different conditions on nano-silica behavior would make the results of the hazard assessment more reliable and minimize the uncertainty associated with the endpoints. Because some of the extrapolations applied to toxicity datasets include the use of acute data to predict chronic toxicity effects (i.e. the acute to chronic ratio). In addition, a low number of endpoint concentrations (only twenty endpoint concentrations were used for the model presented) can be another source of uncertainty for the hazard assessment results.

By using probability distributions instead of single endpoint value, we attempted to address the uncertainty which is inherent in toxicity studies. The hazard assessment we performed is for a "generic" nano-silica and is considering all different sizes, coatings and synthesis methods, representing all types of currently used nano-silica in toxicity studies. This enables us to compare in a next step the "generic nano-silica SSD" with the modeled flows and concentrations which are also for a "generic nano-silica" because data on production and use of specific forms of nano-silica is not available.

Nanoscale materials including silica occur naturally in the atmosphere, in natural waters, the ocean and in biological systems [139]. Conventional materials may also contain a non-negligible fraction of nano-size particles [140, 141]. Amorphous silica can be found in a range of products including toothpaste, cosmetics, paints, foods and dietary supplements [142]. Dissolved silica concentrations in European rivers are in average 5.6 mg/l [143]. Suspended solids in rivers are in average 55% SiO_2 and at a global scale the dissolved/particulate silica ratio is 4.2% [143]. Hence the background concentration of particulate silica in river waters is in the 100 mg/l range. The predicted concentration of nano-silica in surface water in EU is 0.12 μg/l, negligible compared with total particulate silica. Particulate silica is a source of dissolved silica which is an important nutrient and the limiting factor in many aquatic systems. Soluble silicates from industrial applications were modeled to account for only 2% of the dissolved silica in surface water [144]. Dissolved silica is only toxic to aquatic organisms at concentrations well above the solubility limit in the hundreds of mg/l range [144]. The nano-silica flows calculated in our work are therefore neither affecting dissolved nor particulate silica flows in the environment to any significant degree.

There are a number of future steps that could be taken to improve outputs of the environmental risk assessment. First, the further development of the techniques to measure ENMs concentrations in the environment could be used to define the exposure concentration of ENM directly. In addition, a dynamic MFA model, for example, considering dynamic life cycles of products [145], would also help to improve the reliability of the predicted exposure concentrations. In the case of nano-silica, the estimated exposure would be closer to the actual values, if more investigations had been done to achieve better quantitative understanding of transformation processes such as dissolution. Further investigations and data collection are also needed to be able to assess the risk posed by nano-silica in other environmental compartments such as sediments and soil for which currently not enough data is available to perform any reasonable hazard assessment.

A1.5 References

Al-Kattan A, Wichser A, Vonbank R, Brunner S, Ulrich A, Zuin S, et al. Characterization of materials released into water from paint containing nano-SiO2. Chemosphere 2015; 119: 1314-1321.

Amerio E, Fabbri P, Malucelli G, Messori M, Sangermano M, Taurino R. Scratch resistance of nano-silica reinforced acrylic coatings. Progress in Organic Coatings 2008; 62: 129-133.

Anastasio C, Martin ST. Atmospheric nanoparticles. In: Banfield JF, Navrotsky A, editors. Nanoparticles and the Environment, 2001, pp. 293-349.

ANEC/BEUC. ANEC/BEUC Inventory of Products Claiming to Contain Nanoparticles available on the EU Market: https://nanocolors.wordpress.com/2009/11/22/anecbeuc-inventory-of-products-claiming-to-contain-nanoparticles-available-on-the-eu-market/. 2010.

ANSES. Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire) 2013.

Arvidsson R, Molander S, Sandén BA. Particle Flow Analysis. Journal of Industrial Ecology 2012; 16: 343-351.

Bornhöft NA, Nowack B, Hilty LM. Material flow modelling for environmental exposure assessment – A critical review of four approaches using the comparative implementation of an idealized example. . In: B. Page, Fleischer AG, Göbel J, Wohlgemuth V, editors. EnviroInfo 2013, Proceedings of the 27th Conference on Environmental Informatics, 2013, pp. 379-388.

Bosch A, Maier M, Morfeld P. Nanosilica? Clarifications are necessary! Nanotoxicology 2012; 6: 611-613.

Boxall ABA, Chaudhry Q, Sinclair C, Jones AD, Aitken R, Jefferson B, et al. Current and future predicted environmental exposure to engineered nanoparticles. Central Science Laboratory, Sand Hutton, UK, 2007.

BUND. Nanoproduct Database Available from: http://www.bund.net/. Retrieved 15.03.2015. 2015.

Caballero-Guzman A, Sun T, Nowack B. Flows of engineered nanomaterials through the recycling process in Switzerland. Waste Manag 2015; 36: 33-43.

Chang JS, Chang KLB, Hwang DF, Kong ZL. In vitro cytotoxicitiy of silica nanoparticles at high concentrations strongly depends on the metabolic activity type of the cell line. Environ. Sci. Technol. 2007; 41: 2064-2068.

Coll C, Notter D, Gottschalk F, Sun TY, Som C, Nowack B. Probabilistic environmental risk assessment of five nanomaterials (nano-TiO2, nano-Ag, nano-ZnO, CNT, and Fullerenes). Nanotoxicology, 2015.

Dekkers S, Bouwmeester H, Bos PMJ, Peters RJB, Rietveld AG, Oomen AG. Knowledge gaps in risk assessment of nanosilica in food: evaluation of the dissolution and toxicity of different forms of silica. Nanotoxicology 2013; 7: 367-377.

Dekkers S, Krystek P, Peters RJB, Lankveld DPK, Bokkers BGH, van Hoeven-Arentzen PH, et al. Presence and risks of nanosilica in food products. Nanotoxicology 2011; 5: 393-405.

Dumont E, Johnson AC, Keller VDJ, Williams RJ. Nano silver and nano zinc-oxide in surface waters – Exposure estimation for Europe at high spatial and temporal resolution. Environmental Pollution 2015; 196: 341-349.

Dürr HH, Meybeck M, Hartmann J, Laruelle GG, Roubeix V. Global spatial distribution of natural riverine silica inputs to the coastal zone. Biogeosciences 2011; 8: 597-620.

EC21. Global B2b Market Place Available from: http://manufacturer.ec21.com/nano_silica.html. Retrieved 15.03.2015. 2015.

ECB. Technical Guidance Document on Risk Assessment, European Chemicals Bureau. 2003.

ECHA. Guidance on information requirements and chemical safety assessment, Part E risk characterization. 2008a.

ECHA. Guidance on information requirements and chemical safety assessment. Chapter R.10: Characterisation of dose [concentration]-response for environment. 2008b.

ECHA. Guidance on information requirements and chemical safety assessment: Chapter R.16 Environmental exposure estimation. European Chemicals Agency, Helsinki, Finland, 2012, pp. 136.

EPR. European Patent Office. European Patent Register Available from: http://www.epo.org/index.html. Retrieved 14.03.2015. 2015.

Froggett S, Clancy S, Boverhof D, Canady R. A review and perspective of existing research on the release of nanomaterials from solid nanocomposites. Particle and Fibre Toxicology 2014; 11: 17.

Fujiwara K, Suematsu H, Kiyomiya E, Aoki M, Sato M, Moritoki N. Size-dependent toxicity of silica nano-particles to Chlorella kessleri. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 2008; 43: 1167-1173.

Geranio L, Heuberger M, Nowack B. Behavior of silver nano-textiles during washing. Environ. Sci. Technol. 2009; 43: 8113-8118.

Gottschalk F, Kost E, Nowack B. Engineered nanomaterials (ENM) in waters and soils: a risk quantification based on probabilistic exposure and effect modeling. Environ. Toxicol. Chem. 2013a; 32: 1278–1287.

Gottschalk F, Lassen C, Kjoelholt J, Christensen F, Nowack B. Modeling flows and concentrations of nine engineered nanomaterials in the Danish environment. Int J Environ Res Public Health 2015; 12: 5581-602.

Gottschalk F, Nowack B. A probabilistic method for species sensitivity distributions taking into account the inherent uncertainty and variability of effects to estimate environmental risk. Integrated Environmental Assessment and Management 2013; 9: 79–86.

Gottschalk F, Ort C, Scholz RW, Nowack B. Engineered nanomaterials in rivers--exposure scenarios for Switzerland at high spatial and temporal resolution. Environmental pollution (Barking, Essex : 1987) 2011; 159: 3439-45.

Gottschalk F, Scholz RW, Nowack B. Probabilistic material flow modeling for assessing the environmental exposure to compounds: Methodology and an application to engineered nano-TiO2 particles. Environ. Modeling Software 2010; 25: 320-332.

Gottschalk F, Sonderer T, Scholz RW, Nowack B. Modeled environmental concentrations of engineered nanomaterials (TiO_2, ZnO, Ag, CNT, fullerenes) for different regions. Environ. Sci. Technol. 2009; 43: 9216-9222.

Gottschalk F, Sun TY, Nowack B. Environmental concentrations of engineered nanomaterials: Review of modeling and analytical studies. Environmental Pollution 2013b; 181: 287-300.

Handy RD, van der Kammer F, Lead JR, Hassellöv M, Owen R, Crane M. The ecotoxicology and chemistry of manufactured nanoparticles. Ecotoxicology 2008; 17: 287-314.

Hochella MF, Aruguete D, Kim B, Madden AS. Naturally Occurring Inorganic Nanoparticles: General Assessment and a Global Budget for One of Earth's Last Unexplored Major Geochemical Components. Victoria, Austrilia: Pan Stanford Publishing, 2012.

Jin YH, Lohstreter S, Zhao JXJ. Toxicity of Spherical and Anisotropic Nanosilica. Nanotechnologies for the Life Science 2009; 2: 221-239.

Kaegi R, Ulrich A, Sinnet B, Vonbank R, Wichser A, Zuleeg S, et al. Synthetic TiO_2 nanoparticle emission from exterior facades into the aquatic environment. Environ. Pollut. 2008; 156: 233-239.

Kaiser JP, Diener L, Wick P. Nanoparticles in paints: A new strategy to protect façades and surfaces? Journal of Physics 2013; 429: 10.

Keller A, McFerran S, Lazareva A, Suh S. Global life cycle releases of engineered nanomaterials. Journal of Nanoparticle Research 2013; 15: 1-17.

Keller AA, Lazareva A. Predicted Releases of Engineered Nanomaterials: From Global to Regional to Local. Environmental Science & Technology Letters 2014; 1: 65-70.

Klaine SJ, Koelmans AA, Horne N, Carley S, Handy RD, Kapustka L, et al. Paradigms to assess the environmental impact of manufactured nanomaterials. Environ. Toxicol. Chem. 2012; 31: 3-14.

Krug HF, Wick P. Nanotoxicology: An Interdisciplinary Challenge. Angewandte Chemie-International Edition 2011; 50: 1260-1278.

Liu HH, Cohen Y. Multimedia Environmental Distribution of Engineered Nanomaterials. Environmental Science & Technology 2014; 48: 3281-3292.

Lorenz C, Windler L, Lehmann RP, Schuppler M, Von Goetz N, Hungerbühler K, et al. Characterization of silver release from commercially available functional (nano)textiles. Chemosphere 2012; 89: 817–824.

Martin AD, Quinn KM, Park JH. MCMApack: Markov Chain Monte Carlo in R. Journal of Statistical Software 2011; 42: 21.

Meesters JAJ, Koelmans AA, Quik JTK, Hendriks AJ, van de Meent D. Multimedia Modeling of Engineered Nanoparticles with SimpleBox4nano: Model Definition and Evaluation. Environmental Science & Technology 2014; 48: 5726-5736.

Merget R, Bauer T, Kupper HU, Philippou S, Bauer HD, Breitstadt R, et al. Health hazards due to the inhalation of amorphous silica. Archives of Toxicology 2002; 75: 625-634.

Michel K, Scheel J, Karsten S, Stelter N, Wind T. Risk assessment of amorphous silicon dioxide nanoparticles in a glass cleaner formulation. Nanotoxicology 2013; 7: 974-88.

Mizutani T, Arai K, Miyamoto M, Kimura Y. Application of silica-containing nano-composite emulsion to wall paint: A new environmentally safe paint of high performance. Progress in Organic Coatings 2006; 55: 276-283.

Mueller NC, Nowack B. Exposure modeling of engineered nanoparticles in the environment. Environ Sci Technol 2008; 42: 4447-53.

Nanodatabase. The Nanodatabase http://nanodb.dk/. 2013.

Napierska D, Thomassen LC, Lison D, Martens JA, Hoet PH. The nanosilica hazard: another variable entity. Part Fibre Toxicol 2010; 7: 39.

Neal C, Neal M, Reynolds B, Maberly SC, May L, Ferrier RC, et al. Silicon concentrations in UK surface waters. Journal of Hydrology 2005; 304: 75-93.

Nel A, Xia T, Mädler L, Li N. Toxic potential of materials at the nanolevel. Science 2006; 311: 622-627.

Nguyen T, Pellegrin B, Bernard C, Rabb S, Stuztman P, Gorham JM, et al. Characterization of surface accumulation and release of nanosilica during irradiation of polymer nanocomposites by ultraviolet light. J Nanosci Nanotechnol 2012; 12: 6202-15.

O'Brien N, Cummins E. Nano-Scale Pollutants: Fate in Irish Surface and Drinking Water Regulatory Systems. Human and Ecological Risk Assessment 2010; 16: 847-872.

Praetorius A, Scheringer M, Hungerbühler K. Development of Environmental Fate Models for Engineered Nanoparticles—A Case Study of TiO2 Nanoparticles in the Rhine River. Environmental Science & Technology 2012; 46: 6705-6713.

Pronk MEJ, Wijnhoven SWP, Bleeker EAJ, Heugens EHW, Peijnenburg WJGM, Luttik R, et al. Nanomaterials under REACH. Nanosilver as a case study. RIVM report 601780003, 76 pp. . 2009.

Salata OV. Applications of nanoparticles in biology and medicine. Nanobiotechnology 2004; 2: 6.

SASSI. Nanoscale Material Stewardship Programm (NMSP) Voluntary Submittal Package for Sythetic Amorphous Silica (CAS NO. 7631-86-9). . 2008.

Sun TY, Gottschalk F, Hungerbühler K, Nowack B. Comprehensive Modeling of Environmental Emissions of Engineered Nanomaterials. Environ. Pollut. 2014; 185: 69-76.

USP&TO. United States Patent and Trademark Office. Patent Full-Text Databases Available from: http://patft.uspto.gov/netahtml/PTO/index.html. Retrieved 14.03.2015. 2015.

Van Hoecke K, De Schamphelaere KA, Ramirez-Garcia S, Van der Meeren P, Smagghe G, Janssen CR. Influence of alumina coating on characteristics and effects of SiO2 nanoparticles in algal growth inhibition assays at various pH and organic matter contents. Environ Int 2011; 37: 1118-25.

Van Hoecke K, De Schamphelaere KAC, Van der Meeren P, Lucas S, Janssen CR. Ecotoxicity of silica nanoparticles to the green alga Pseudokirchneriella subcapitata: Importance of surface area. Environmental Toxicology and Chemistry 2008; 27: 1948-1957.

Vance EM, Kuiken T, Vejerano PE, McGinnis PS, Hochella Jr FM, Rejeski D, et al. Nanotechnology in the real world: Redeveloping the nanomaterial consumer products inventory. Beilstein Journal of Nanotechnology 2015; 6: 12.

von der Kammer F, Ferguson PL, Holden PA, Masion A, Rogers KR, Klaine SJ, et al. Analysis of engineered nanomaterials in complex matrices (environment and biota): General considerations and conceptual case studies. Environ. Toxicol. Chem. 2012; 31: 32-49.

Walser T, Limbach LK, Brogioli R, Erismann E, Flamigni L, Hattendorf B, et al. Persistence of engineered nanoparticles in a municipal solid-waste incineration plant. Nat Nano 2012; 7: 520-524.

Wei CX, Zhang YB, Guo J, Han B, Yang X, Yuan JL. Effects of silica nanoparticles on growth and photosynthetic pigment contents of Scenedesmus obliquus. Environmental Science 2010; 22: 6.

Weir A, Westerhoff P, Fabricius L, Hristovski K, von Götz N. Titanium Dioxide Nanoparticles in Food and Personal Care Products. Environmental Science & Technology 2012; 46: 2242-2250.

Wohlleben W, Brill S, Meier MW, Mertler M, Cox G, Hirth S, et al. On the Lifecycle of Nanocomposites: Comparing Released Fragments and their In-Vivo Hazards from Three Release Mechanisms and Four Nanocomposites. Small 2011; 7: 2384–2395.

Yang SY, Ye RF, Han B, Wei CX, Yang X. Ecotoxicological Effect of Nano-silicon Dioxide Particles on Daphnia Magna. Integrated Ferroelectrics 2014; 154: 64-72.

Zuin S, Gaiani M, Ferrari A, Golanski L. Leaching of nanoparticles from experimental water-borne paints under laboratory test conditions. J Nanopart Res 2014; 16: 2185.

Annex 2. Are engineered nano iron oxide particles safe? An environmental risk assess-ment by probabilistic exposure, effects and risk modelling

Nanotoxicology 2016, volume 10, page 1545-1554

Abstract

Nano iron oxide particles are beneficial to our daily lives through their use in paints, construction materials, biomedical imaging and other industrial fields. However, little is known about the possible risks associated with the current exposure level of engineered nano iron oxides (nano-FeOX) to organisms in the environment. The goal of this study was to predict the release of nano-FeOX to the environment and assess their risks for surface waters in the EU and Switzerland. The material flows of nano-FeOX to technical compartments (waste incineration and waste water treatment plants) and to the environment were calculated with a probabilistic modeling approach. The mean value of the predicted environmental concentrations (PECs) of nano-FeOX in surface waters in the EU for a worst-case scenario (no particle sedimentation) was estimated to be 28 ng/l. Using a probabilistic species sensitivity distribution, the predicted no effect concentration (PNEC) was determined from ecotoxicological data. The risk characterization ratio, calculated by dividing the PEC by PNEC values, was used to characterize the risks. The mean risk characterization ratio was predicted to be several orders of magnitude smaller than 1 (0.00014). Therefore, this modeling effort indicates that only a very limited risk is posed by the current release level of nano-FeOX to organisms in surface waters. However, a better understanding of the hazards of nano-FeOX to the organisms in other ecosystems (such as sediment) needs to be assessed to determine the overall risk of these particles to the environment.

A2.1 Introduction

Assessing the environmental risks of engineered nanomaterials (ENMs) has received a lot of attention in the last years. Whereas for ENMs such as nano-Ag and nano-TiO_2 a large amount of research on exposure, hazard and risk has been published, other materials have only received very limited attention despite potentially high poduction amounts. An example of these under-studied materials is engineered nano-iron oxides (nano-FeOX). Iron oxides are not only engineered materials, but are also present in various forms in natural systems, both in bulk as well as in nano-form [146]. Nano-scale iron oxides can be formed under mineral weathering process and biotic processes or nanobiomineralization [147-149]. Naturally occuring nano-scaled iron oxides are widely distributed throughtout the atomosphere, ocean, water, and most living organisms [150].

Engineered nano-FeOX have unique properties that can be applied in various industrial fields. Nano-FeOX can be fully dispersed in a matrix due to the small size and do not scatter light anymore. Therefore, they can be used to manufacturing transparent nano-FeOX transparent coatings with high UV absorption and increased durability [69]. Nano-FeOX based materials have been found to be effective catalysts involded in oxygen evolution processes, such as water splitting and chloring evolution [71, 151, 152]. Nano-sized FeOX particles are more efficient in catalysis than micro-sized particles [73]. Magnetic nano-FeOX can be used in both in vivo and in vitro biomedical application. e.g. drug targeting, hyperthermia and nuclear magnetic resonance imaging, [74-76, 153]. With their large surface area and adjustable surface charge, nano-FeOX can be used to absorb cations such as Cd, Co, Zn etc. and anions such as PO_4^{3-} [150, 154, 155]. Moreover, when the particle size is below a few nanometers, nano-FeOX become superparamagnetic, which facilitates phase separation processes after metal binding and selective adsorption [156, 157].

Due to these diverse uses and an increasing market demand, nano-FeOX can be expected to be released to the environment. However, little is known regarding the environmental exposure and risk of nano-FeOX . It is therefore imperative to quantify the release of nano-FeOX to natural systems and determine their possible risks on organisms.

To assess the release of nano-FeOX to the environment from applications, a modeling appraoch is currently the only choice because analytical methods are not advanced enough to distinguish between engineered and naturally occurring nanoparticles [33]. Exposure models for ENMs have evolved since Mueller and Nowack initially set the scene with a first material flow modeling framework [35]. The model was updated with a lifecycle concept to trace ENMs from cradle to grave and a probabilistic approach to address the uncertainty of the input parameters [36, 37]. Material flow models have been applied to calculate the mass flows and concentrations for various ENMs in different regions [41, 42, 158, 159]. Keller et al. (2013) predicted the combined flows of nano-FeOX /zero-valent iron (NZVI) to landfill, air, water and soil at a global level, where seven industrial sectors were considered [39]. In a follow-up study in 2014, the same authors calculated the released amount of nano-FeOX /NZVI for eight different regions and predicted the concentration of nano-FeOX /NZVI in wastewater treatment plant effluents and bio-solids in California [40]. The models in the two aforementioned studies used mathematic algorithms, in which the input parameters are based on data from one market research report (production volume and distribution to products of nano-FeOX /NZVI).

The toxic effects of nano-FeOX have been studied including short-term and long-term toxicity [160-166]. Early toxicity studies for nano-FeOX found that their toxicity is low and that there is no link between size and toxicity [167, 168]. However, some later studies found that nano-FeOX can be cytotoxic and can induce systemic toxicity [164, 169]. Prodan et al. (2013) have shown that nano-FeOX can be toxic to rats and a size-dependent toxicity has been found by Hazeem et al. (2015). Zhang et al. (2015) observed a high accumulation of nano-FeOX in Zebrafish, accompanied, however, by a high elimination rate.

In environmental risk assessment, the risk is quantified by comparing the predicted environmental concentration (PEC) and the predicted no effect concentration (PNEC) [60]. The PNEC represents the critical concentration at which no effects are expected in a given ecosystem. The PNEC can be obtained from the lowest observed no-effect concentration (NOEC) or from statistical extrapolation methods using species sensitivity distribution (SSD) [60]. SSDs have been built for various ENMs [54, 57, 58]. However, up to date, no evaluation using SSD has been performed so far for nano-FeOX.

NZVI and nano-FeOX have common properties (e.g. magnetic properties) and similar potential for environmental applications and therefore they are often grouped together, e.g. in [39]. NZVI has emerged as a new option in treating soil and groundwater with contaminants [171]. Due to its susceptibility to oxidation, it is rapidly oxidized to form an iron oxide when oxygen is present. To estimate the environmental risk of nano-FeOX more precisely, in the current paper the target material is nano-FeOX only, not including NZVI or its transformation products.

To date, no complete environmental risk assessment for nano-FeOX has been conducted. Therefore, this study aims to assess the release of nano-FeOX to the EU and Swiss environment, predict exposure levels and assess the risks of nano-FeOX to the environment using a probabilistic material flow modeling and risk assessment approach.

A2.2 Methods

A2.2.1 Model description

A material flow modeling approach was used to predict the mass flows and environmental concentrations of nano-FeOX for the targeted environmental compartments, including air, water, soil, and sediments as shown in Figure A2.1. The targeted system boundaries in this study are EU and Switzerland (CH). The model is based on a material flow analysis from a life cycle perspective to analyze mass flows of nano-FeOX to the environment as it can be released through the whole lifetime [37, 41, 158]. The model includes four different stages: production of nanoparticles; manufacturing and consumption of consumer products incorporating nano-FeOX ; waste management processes; and environmental compartments. Mass loss during production and manufacturing processes were considered in the model, as described in Mueller and Nowack (2008) . Waste management compartments include landfill, waste incineration plant (WIP), wastewater treatment plant (WWTP), recycling and export. Potential leaching from landfills was not included in this iteration of the model. However, the behavior of nano-FeOX after it flows into the recycling processes was considered based on the study from Caballero-Guzman et al. (2015). However, this investigation was only done for four product categories: 'Electronics'; 'Paints'; 'Battery'; and 'Construction' because no data were reported for the other product categories in [132]. As the basis for the previous recycling model only targeted the Swiss system, we extrapolated this data and assumed that the fate of nano-FeOX during recycling is the same in the EU as in Switzerland.

Environmental compartments considered in the modeling were air, soil, surface water, and sediment. In the case of the EU, the soil compartment was separated into 'natural & urban soil' (N&U soil) and 'sludge treated soil' (ST soil) because sewage sludge is applied as fertilizer in most European countries. The retention time of nanoparticles in the air and surface water system were defined as 10 and 40 days, respectively [110, 111]. Two worst-case scenarios were chosen for the aquatic system: no sedimentation for the surface water compartment and complete sedimentation for the sediment compartment. It was assumed that all compartments are homogenous and well-mixed systems. Based on the model framework, three fundamental input parameters are required: production volume, product allocation, and transfer factors among different compartments.

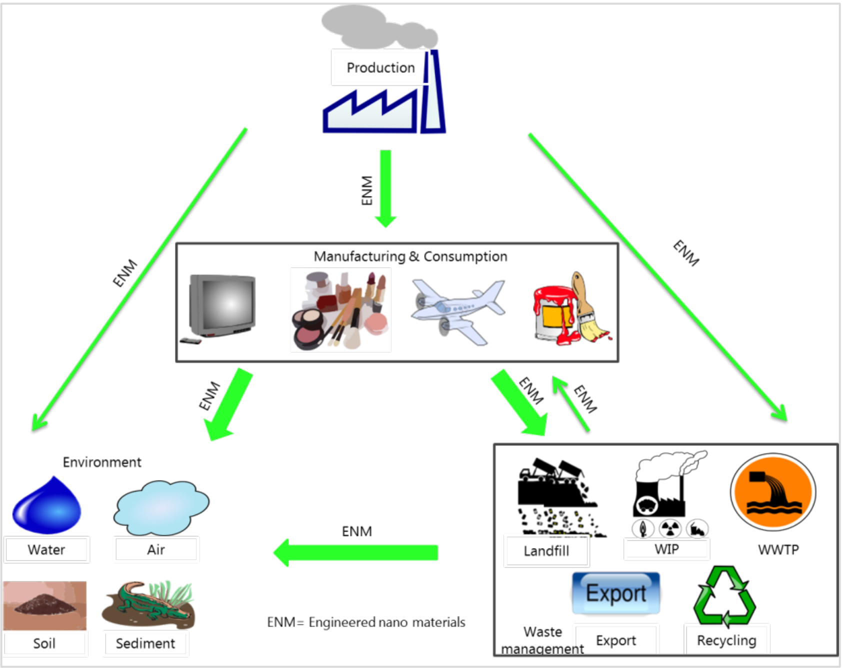


Figure A2.1. Framework of the material flow model. The framework contains four parts: production; manufacturing & consumption; waste management; and the environment. Waste management includes landfills, waste incineration plants (WIP), waste water treatment plants (WWTP), recycling and export. There are four environmental compartments: air, water, soil and sediment. The arrow from the waste management box to manufacturing represents reuse of materials after recycling.

A2.2.2 Input parameters

A2.2.2.1 Production volume

The goal was to predict the material flows and risk of nano-FeOX in the environment in the calendar year of 2014. The most fundamental input parameter of the model is the production volume. Data regarding produced amounts of nano-FeOX was collected from different sources such as market research reports and official registration reports, reflecting the production values for the year 2014. All production data of years other than 2014 were projected to 2014 based on a published general development of ENMs production over time [46] . We assumed that the development of production volume of nano-FeOX was proportional to the general development of ENMs production as no specific data for nano-FeOX are available. The ratios between each year to the reference year 2014 were brought into the model as a probability distribution, presenting a comprehensive picture of the real production volume for the calculated year compared to the reference year 2014. All data were extrapolated to estimate the production volume in the EU and Switzerland using GDP ppp (Gross Domestic Product per capita based on Purchasing Power Parity) [172].

To handle the uncertainty in production volumes from various datasets, each value was introduced to the model as a probability distribution [41, 158]. Moreover, a Degree of Belief (DoB), 20 or 80, was assigned to each source as a measure of reliability. The detailed explanation and the decision tree of choosing a certain DoB is described in supporting information. The DoB of a data source was reflected by the sample size during the modeling. The sample size of each input parameter is 100,000. Hence the sampling sizes were for example 20,000 for data sources with DoB of 20 and 80,000 for those with DoB of 80.

A2.2.2.2 Product allocation

The distribution of nano-FeOX to nano-products is the second most important input parameter for our modeled system. This allocation determines the further fate of nano-FeOX based on the life cycles of different product categories. Because no single database provides a comprehensive overview of products containing nano-FeOX , a list of categories for nano-FeOX was defined based on market reports and the official French registration report [64, 129, 173]. Four different methods were used to generate the share of nano-FeOX in different categories: M1- using the data in the Future Markets report from 2012 to calculate the allocation of nano-FeOX in each category [63]; M2- a patent search of U.S. and European patent inventories [116, 117]; M3- using the known market value of each product category incorporating nano-FeOX to calculate the product distribution; and M4- taking available direct production data for nano-FeOX in a product category. A DoB was assigned to each method depending on their reliability. A detailed description of the four methods and the data collected is described in the Table A2.2.

A2.2.2.3 Transfer coefficients

Transfer coefficients determine how much nano-FeOX flows between different compartments of the model. However, there is no study available regarding releases of nano-FeOX from commercial applications. The transfer factors were therefore based on those of the same product categories containing engineered nano-TiO_2 from Sun et al. (2014). This is because nano-FeOX is used in the same or similar products as engineered nano-TiO_2 such as paints, electronics and cosmetics, which have the same or similar lifecycles. Table A2.3 presents where nano-FeOX goes during use and disposal. WWTP and WIP are technical processes where significant amounts of nano-FeOX are captured. Therefore, the removal efficiencies of these processes are critical for the environmental releases of nano-FeOX . Nano-FeOX share similar properties with other nano metal oxides. e.g. solubility and reactivity. No information is available regarding removal efficiency of nano-FeOX during wastewater treatment or WIP. We therefore used the removal rates of total iron and nano-TiO_2 (Table S2.4) to build a probability distribution (Figure S2.2) for nano-FeOX . For the removal rate in WIP, the data for engineered nano-CeO_2 from a full-scale WIP was chosen [122]. The transfer coefficients within the recycling systems are chosen from Caballero-Guzman et al. (2015).

A3.2.3 Hazard assessment

For the hazard assessment, the potential hazardous effects posed by nano-FeOX to organisms were evaluated and PNECs for a specific environmental system were obtained. The collected endpoint concentrations (EC50, EC10, EC55, LC50, NOEC and LOEC) contains ecotoxicological data of both modified and non-modified nano-FeOX. All sensitivity concentrations were converted to NOEC values using assessment factors. The criteria for choosing the assessment factors were described in our previous study [172]. We generated the PNEC values from the 5th percentile of cumulative SSD as recommended by the REACH guidance [174]. In this study we combined individual probabilistic species sensitivity distributions (PSSD) of each species to generate one PSSD [54, 172]. With 100’000 simulation runs, a PNEC distribution was derived by abstracting the 5th percentiles from PSSDs [55]. Due to the limited availability of ecotoxicological data, the risk assessment for nano-FeOX is currently only feasible for surface waters.

A3.2.4 Risk assessment

The environmental risk was quantified by comparing the PEC with the PNEC values. The risk characterization ratio (RCR), the ratio of PEC over PNEC shown in equation 1, was used as a measure of the risk level [52]. Here, if the RCR value is over 1, it indicates that the risk management measures should be taken regarding a given material. If not, it suggests the risk is under control at the current release level.

$$\mathrm{RCR} = \mathrm{PEC}/\mathrm{PNEC} \qquad (1)$$

Instead of calculating one single RCR, a RCR distribution was derived by dividing all PEC values by all PNEC values of the respective probability distributions [55].

A2.3 Results

A2.3.1 Exposure assessment

A2.3.1.1 Nano-FeOX production values

The data sources used to obtain production volume of nano-FeOX with their assigned DoBs are shown in Table S2.1. The values are valid for different years and are extrapolated to 2014 using the scaling factors from Sun et al. (2016). The summary of the production volumes for the EU and Switzerland in 2014 is presented in Table A2.1. The modeled mean production amount is 7,500 tons/year for the EU, with a range of the 15 to 85% percentiles from 2.9 to 9400 tons/year. The reported production numbers cover a very large range, indicating a high uncertainty about the amount produced. The scaled mean production for Switzerland is 190 tons/year. Because no information shows if the produced materials are all used up within the reference year or stored for next year onwards, we equate production to use as done in all previous studies about material flows of ENMs.

Table A2.1. Summary of production volumes in the EU and Switzerland in 2014 (Tons)

System	Q15	Mean	Median	Q85
EU	2.9	7,500	910	9,400
CH	0.060	190	35	230

A2.3.1.2 Product allocation

14 product categories were identified based on the information collected using the 4 different methods. Table A2.2 shows the share of nano-FeOX in each product category calculated by the different methods, as well as the assigned DoB of each method. The weighted mean values of the shares of nano-FeOX in each category from all methods were calculated using the DoB and they were included in the model as triangular distributions. The 9 major categories with the share of nano-FeOX used in each category are displayed in Figure A2.2. The category "Others" expresses those categories where less than 1% of nano-FeOX is used. 37.6% of total nano-FeOX is applied in 'paints' followed by the categories 'Electronics' and 'Construction', which account for 16% and 12.9% respectively. These estimations are consistent with the Future Markets report (2014), which is primarily based on interviews. In the report it desciribes high volume applications are pigments, polishing, and electronics. A significant amount of nano-FeOX also is allocated to the categories 'Cosmetics' and 'Catalyst'. Smaller fractions of nano-FeOX are used in categories such as batteries, water treatment and plastics.

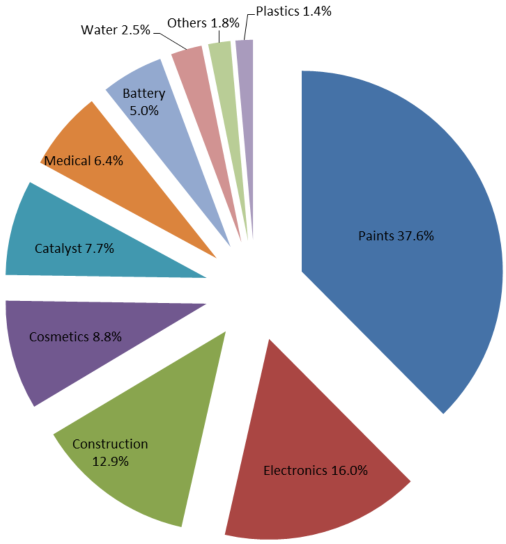


Figure A2.2. Distribution of nano-FeOX in product categories

A2.3.1.3 Mass Flows of nano-FeOX

Before calculating the mass flow to different compartments, transfer factors were assessed (Table S2.3). The data collected for removal efficiency during wastewater treatment are shown in Table S2.4, the resulting probability distribution if the transfer factor in Figure S2.2.

Based on the input parameters mentioned above, probability distributions for each flow were generated. The median, mean, Q15 and Q85 values were abstracted from each distribution and are given in Table A2.5. Figure A2.3 illustrates the mean mass flows, giving an overview of the mass distribution of nano-FeOX among all compartments in the EU and Switzerland. The only inflow is from 'production, manufacturing and consumption'. Nano-FeOX flows out of the system boundary through 'export' or 'recycling'. This is because some wastes in the EU and Switzerland are exported to other counties to be reused or recycled (e.g. E-waste). The most dominant flow is from "Production, manufacturing and consumption (PMC)" to recycling. 3,100 tons nano-FeOX in the EU goes to recycling processes and this number for Switzerland is 75 tons. Another significant flow is from 'PMC' to 'landfill', in the EU, 1,700 out of total 7,500 tons of produced nano-FeOX (75 out of 190 tons in Switzerland). The main flows to surface water are from the 'PMC' compartment, the untreated wastewater and from the effluent from wastewater treatment plants. The amount of nano-FeOX released to the air is very small and is mostly from the PMC compartment. The nanoparticles in air further deposit to the soil and surface water. Soils and sediment were compartments where nano-FeOX is likely to accumulate. The main input to soil is through application of sewage sludge. However, in Switzerland, sludge is incinerated or used as alternative fuel in cement plants and therefore the overall flow of nano-FeOX to soil is much smaller than in the EU.

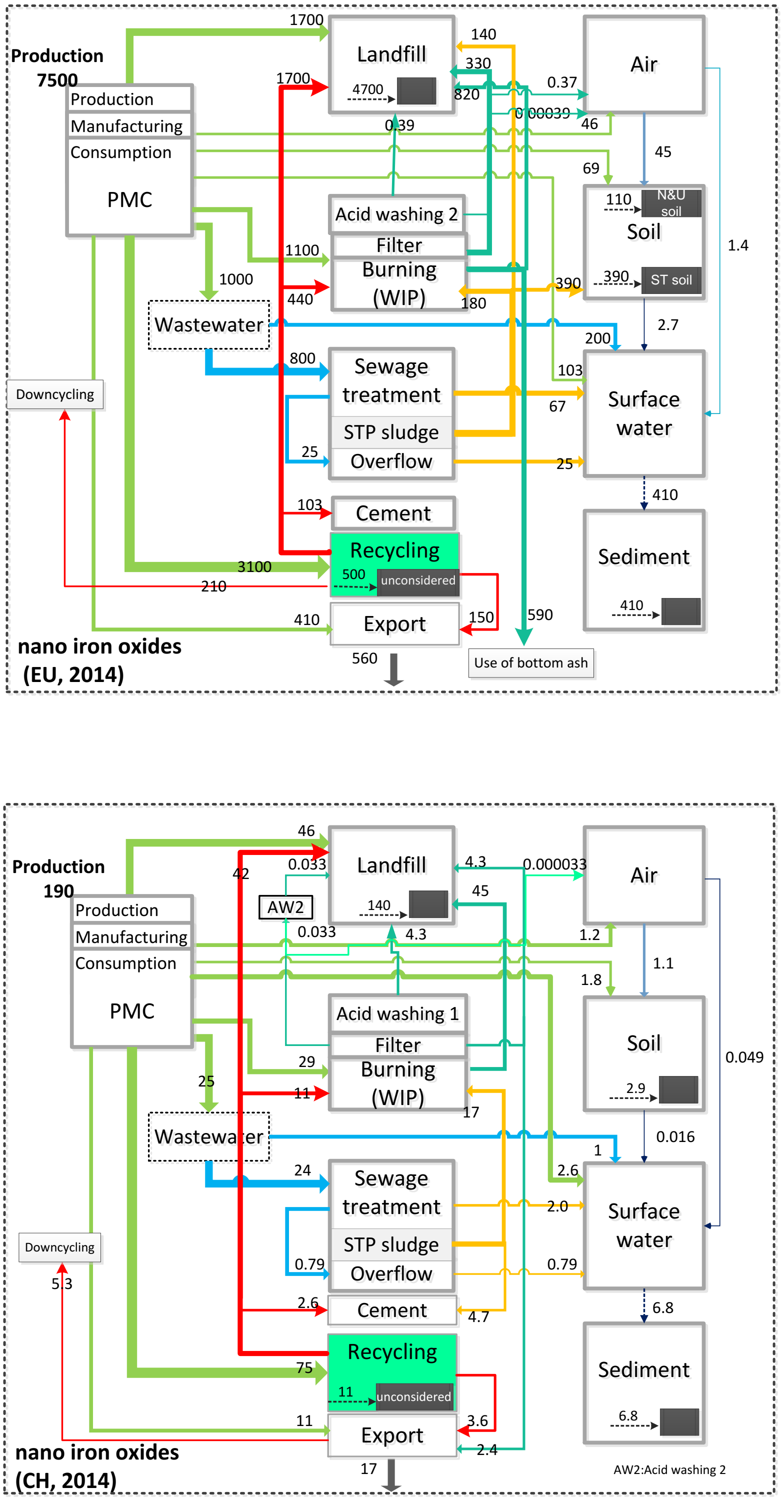


Figure A2.3. Material flows of nano-FeOX in the EU (top) and CH (bottom) in tons/year. Nano-FeOX flows out of the same compartment are shown in the same color. All values are rounded to two significant numbers. The input and output flows are balanced in terms of mean values of the probabilistic distribution. 'Unconsidered' represents product categories which were not considered in the recycling process evaluation of Caballero-Guzman et al. (2015).

The fate of nano-FeOX during recycling processes was also included in the model (Table S2.5). After entering the recycling process, nano-FeOX flows to ‘Cement’, ‘WIP’, ‘Export’, ‘PMC’, and ‘Landfill’. Nano-FeOX flows into the recycling compartment from nine product categories, but due to limited data availability we could only include the further fate of nano-FeOX contained in ‘Paints’, ‘Construction’, ‘Electronics’ and ‘Battery’ [132]. A significant percentage of nano-FeOX (2,600 out of 3,100 tons in the EU, 64 out of 75 tons in Switzerland) went to recycling and could be further allocated to outgoing flows. The remaining mass was attributed to the "unconsidered" sub-compartment and not analyzed any further. A major fraction of the recycled materials containing nano-FeOX was landfilled (1700 tons in the EU and 42 tons in Switzerland). In addition, a noticeable flow from recycling processes goes to waste incineration plants (440 out of 2600 tons in the EU). It is obvious from the flow charts that differences in the waste management process exist between the EU and Switzerland that have a profound influence on the final mass flows, e.g. sludge from the WWTP is incinerated in Switzerland (78%), while it can be landfilled (20%) or applied as fertilizer in the EU (55%).

A2.3.1.4 Predicted nano-FeOX concentrations in technical and environmental compartments

The predicted nano-FeOX concentrations were calculated by dividing the mass flows by the mass or volume of each compartment, as described in further detail in Sun et al. (2014). These data therefore provide average concentrations in well-mixed systems. Environmental concentrations were estimated for four environment compartments: air, soil (natural and urban soil (N&U soil) and sludge-treated soil (ST soil)), surface water, and sediment; in addition, concentrations of nano-FeOX in effluent and sludge from wastewater treatment plants were calculated, as well as concentration of nano-FeOX in the waste, WIP fly ash, and WIP bottom ash. Table A2.3 presents the mean, median, Q15, and Q85 values of the predicted concentrations. The concentrations for soils and sediments correspond to the yearly increases in concentrations since the ENMs are assumed to accumulate there. The lowest concentration is found in air (EU, 0.29 ng/m^3 and CH, 0.76 ng/m^3). This is because the application of nano-FeOX in products results only in small amount of releases to the atmosphere. In the complete sedimentation scenario, the concentration in sediments is predicted to be the highest of all environmental concentrations (EU, 390 ug/kg·y and CH, 450 ug/kg·y). However, our model is a static model and therefore the accumulation of nano-FeOX in all compartments from previous years was not considered. The calculated concentrations are only based on the input flow in 2014.

Table A2.3. Concentrations of nano-FeOX in technical and environmental compartments. For soil and sediments the yearly increase in concentration is given. All values are rounded to two significant numbers.

EU					
	Q15	**Mean**	**Median**	**Q85**	**Unit**
Air	1.1E-04	2.9E-01	3.4E-02	3.6E-01	ng/m3
Sediment	1.6E-01	390	44	490	µg/kg·y
N&U* soil	3.8E-05	9.4E-02	1.1E-02	1.2E-01	µg/kg·y
ST* soil	4.4E-02	110	13	140	µg/kg·y
Surface water	1.1E-02	2.9E+01	3.4	36	ng/L
Waste	1.2E-02	31	3.7	39	mg/kg
WIP* bottom ash	7.0E-02	191	22	242	mg/kg
WIP* fly ash	1.0E-01	261	29	334	mg/kg
WWTP* effluent	4.9E-04	2.2	1.2E-01	2.4	µg/L
WWTP* sludge	3.1E-02	79	9.3E+00	100	mg/kg
CH					
	Q15	**Mean**	**Median**	**Q85**	**Unit**
Air	2.7E-04	7.6E-01	9.4E-02	9.7E-01	ng/m3
Sediment	1.5E-01	450	50	560	µg/kg·y
N&U* soil	9.2E-05	2.5E-01	2.9E-02	3.3E-01	µg/kg·y
ST* soil	0	0	0	0	µg/kg·y
Surface water	2.0E-02	5.5E+01	6.7	69	ng/L
Waste	2.4E-02	69	8.5	87	mg/kg
WIP* bottom ash	4.0E-02	120	14	150	mg/kg
WIP* fly ash	5.0E-02	140	16	180	mg/kg
WWTP* effluent	6.6E-04	3.4	1.9E-01	3.6	µg/L
WWTP* sludge	3.7E-02	110	12	140	mg/kg

** N&U soil: Natural and urban soil; ST soil: Sludge treated soil; WIP: Waste incineration plant; WWTP: Wastewater treatment plant*

A2.3.2 Hazard assessment

The hazard assessment could only be conducted for the surface water compartment because insufficient ecotoxicity studies with usable endpoint concentrations for soils and sediments were available. From freshwater studies, 26 endpoint concentrations were collected from 12 species from unicellular, invertebrate, and vertebrate groups (Table A2.4). The original test concentration, assessment factors and the calculated NOEC values are displayed in Table S1.6.

Table A2.4. Test species and taxonomic groups, and total number of endpoints

Taxonomic group	**Test organism**	**Total number of endpoints available**
Unicellular	*V. fischeri, E. coli, P. phosphoreum, R. subcapita*	9
Invertebrate	*D. magna, B. plicatilis, P. subcapita,* *C. dubia, D. magna, T. thermophila*	11
Vertebrate	*D. rerio, O. latipes,*	6

Figure A2.4 shows the NOEC values considered in this work and the cumulative SSD. The most sensitive species from the data points collected is a unicellular organism, *T. thermophila (Tetrahymena thermophila). C. dubia (Ceriodaphnia dubia)* is the least affected species by nano-FeOX in surface water from the species studied thus far. The NOEC values of *Daphnia magna* differ by two orders of magnitude (from 23 µg/l to 5000 µg/l), a range that is smaller than that for other ENMs (where the data for one species can have a range of up to five orders of magnitude) [55].

The probability distribution of the PNEC was built based on the 5th quantile from each PSSD generated during the simulation run. The obtained mean value of the PNEC is 218 µg/l, with a range from 169 µg/l (Q15) to 267 µg/l (Q85). A sensitivity analysis was conducted to evaluate the influence of the lowest endpoint concentration to *D. magna* on the PNEC value. A new PSSD, obtained by removing the lowest NOEC value for *D. magna* is given in Figure S2.3. The new PNEC value was calculated to be 239 µg/l, an increase by 8% compared to the PNEC from the complete data set.

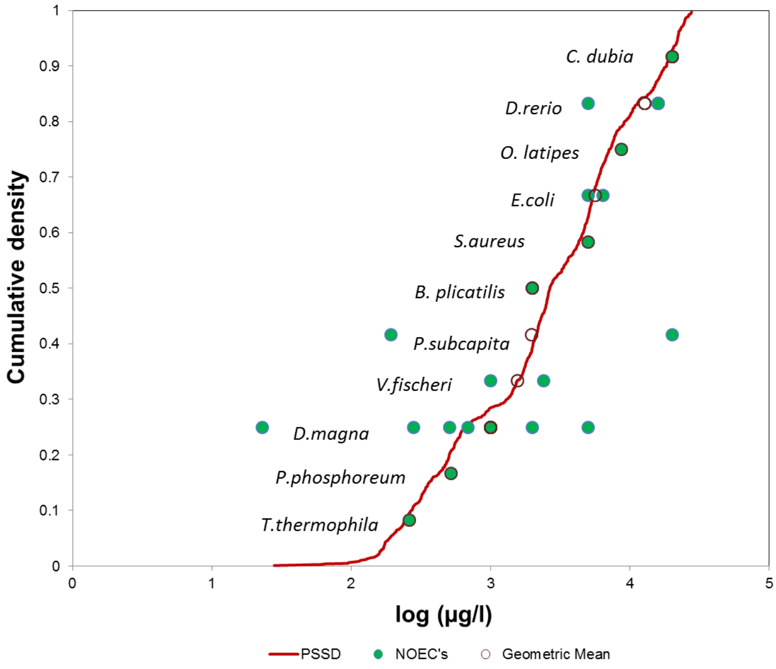


Figure A2.4. Cumulative PSSD for nano iron oxides in the surface water compartment: the red curve is the cumulative PSSD; green dots are the NOEC values, the empty circles are the geometric mean of NOEC for each species.

A2.3.3 Risk characterization

The distributions of PEC and PNEC are plotted together in Figure A2.5 for the EU. The corresponding plot for Switzerland is shown in Figure S2.4. This plot shows that the expected environmental concentrations are several orders of magnitude below the concentrations where effects can be expected.

The risk characterization ratios were calculated by dividing all values of the PEC distribution by those of the PNEC distribution. The mean values of the RCR are 1.4 x 10^{-4} for the EU and 2.6 x 10^{-4} for Switzerland. The likely range of the RCR is from 5.3 x 10^{-8} (Q15) to 1.7 x 10^{-4} (Q85) for the EU (for CH from 9.3 x 10^{-8} to 3.3 x 10^{-4}) which indicates that the potential risk posed by nano-FeOX in the EU and Swiss surface water systems is very limited. The complete RCR distribution curves are given in Figure A2.5.

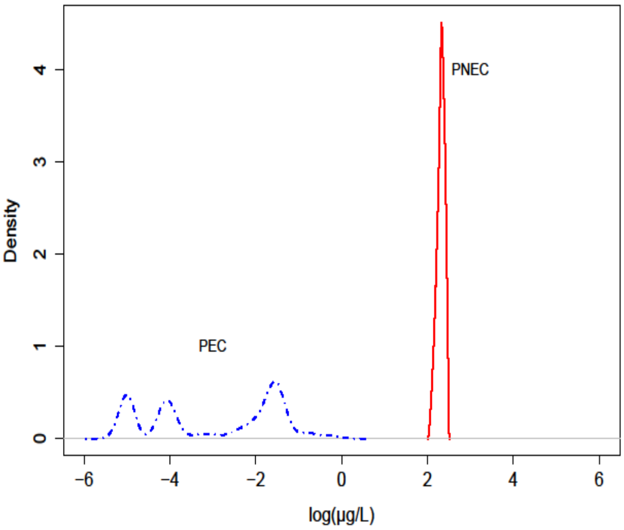


Figure A2.5. The predicted no-effect concentration (PNEC) distribution (in red) compared with the probability density distribution of predicted environmental concentrations (PEC) (in blue) for nano-FeOX in the European surface water system.

A2.4 Discussion

This work provides the first probabilistic flow assessment and the first risk assessment for nano-FeOX in aquatic systems. Many steps in this risk assesment process are based on data with high variability or uncertainty and we discuss in the following the various parts of the modeling in this context.

In the current situation with limited knowledge about the fate of ENMs applied in commercial products, all estimations of environmental flows and risks are definitely associated with uncertainties and variability. One of the main causes of uncertainty in the flow modeling presented here is the overall production amount of nano-FeOX . Data regarding production collected from different sources varies widely. The highest production volumes were obtained from Schmid and Riediker (2008) (scaled to the EU and CH: 32,000 and 365 tons/a), which were about 10,000 and 5,000 times more than the production volumes provided by the Future Markets report (2014) (scaled to the EU and CH: 2.7 and

0.0675 tons/a). The Schmid and Riediker value was derived from a mandatory survey of all companies in Switzerland using nanomaterials and therefore constituted a source with high credibility. The Future Markets values are the most recent and come from a source with an in-depth evaluation of production and use of nanomaterials. The variability of the production volume is likely caused by different perspectives of defining a nanomaterial. EU member states, agencies and economic operators are recommended to use the nanomaterial definition provided by the EU Commission [175]. Studies using other or not clear definitions before 2011 may consider different types of materials as nanomaterials (e.g. including or excluding some traditional materials with similar size distribution). Older data were therefore assigned a lower DoB and thus had less influence on the final result than more recent data.

In all mass flow analyses for ENMs that have been performed so far, product allocation is the weakest step in the modeling process, potentially having a large influence on the final mass flows because the life cycles of products are decisive in determining the potential and the magnitude of release. Without exception, another origin of uncertainty of the mass flows of nano-FeOX is the uncertain allocation of the produced nano-FeOX to different product categories. The properties and life-cycles of the products determine where and how much of nano-FeOX is released. In this study, the product categories were defined mainly based on the French registration report, supported by a survey and market research reports [64, 176, 177]. The share of nano-FeOX in each category was determined by four different methods. With this approach we aimed to capture all possible products and applications that are on the market. Certain products only appeared in the result of one method whereas others came out in all of the 4 methods. These 'certain' product categories such as paints and electronics also have the major market share and therefore dominated the overall flows of nano-FeOX . A product category only covered by one evaluation method makes up a small share of the total mass and it only had a small influence on the overall share of nano-FeOX in this category. The use of different methods therefore increased our confidence in the product allocation obtained.

Transfer factors decide where nano-FeOX distributes after the nano-enabled products were used. The dominant transfer factors are the release fractions from a product to technical and environmental compartments. Other important factors are the removal efficiency during waste management processes. Release from nano-FeOX enabled products, in most cases, is not nano specific, but depends on the life cycles of the products and we have therefore used in most cases the transfer factors for nano-TiO_2 also for nano-FeOX . This is justifiable since both engineered nano-TiO_2 and nano-FeOX are rather insoluble and have similar chemical and physical properties. Therefore, it is possible to use previous release assessments of products containing other ENMs for nano-FeOX -enabled products. Some of transfer factors of nano-FeOX in this work were based on surveys, e.g. for 'Paints' and 'Construction' [178]. Other transfer coefficients were chosen from the transfer factors of engineered nano-TiO_2 for the same product categories or the categories with similar life cycles from Sun et al. (2014).

The flow analysis revealed that nearly 40% of nano-FeOX was discharged to recycling in both the EU and Switzerland. This can be explained by the fact that a significant amount of nano-FeOX is used in ‘electronics’, ‘paints’, and ‘construction’, which were recycled in significant amounts as e-waste and construction waste. From 'production/manufacturing/use', approximately 23% of nano-FeOX flows to landfill, 13% deposits in wastewater, 15% in waste incineration and approximately 5% of nano-FeOX flowed out of the system boundaries since the nano-enabled products were exported. The rest of nano-FeOX was distributed to air, soil, and surface water from ‘PMC’.

As 40% of nano-FeOX flows to the recycling system, it was important to further investigate the fate of nano-FeOX during the recycling processes. Referring to the previous study by Caballero-Guzman et al. (2015), 4 categories (over 80% of nano-FeOX) was considered after entering the recycling stage. The majority of the recycled nano-FeOX was predicted to be landfilled (65%) and WIP (16%), which is a consequence of the use of over 50% of nano-FeOX in the ‘Paints’ and ‘Construction’ categories, which have a high recycling rate of 46% and 70%, respectively (in the EU). Around 8% of recycled nano-FeOX from the ‘Paints’ and ‘Construction’ categories was reused in cement plants and approximately the same amount was reused in other materials coming from construction waste recycling. Nano-FeOX within the product categories ‘Electronics’ and ‘Battery’ was also exported as E-waste out of Europe and Switzerland to be reused or recycled. To improve the current model, more studies are needed to investigate specific fate of nano-FeOX during recycling processes.

The distribution of nano-FeOX to product categories with different life cycles strongly affects the mass of nano-FeOX flows to environmental compartments, as well as environmental concentrations. The predicted concentrations of nano-FeOX were calculated to be 0.011-36 ng/l in European surface water. The concentration range for engineered nano-TiO_2 in Sun et al. (2014) in the EU was estimated to be 0.4-1.4 μg/l, the upper boundary was 38 times larger than that of nano-FeOX . However, the input flow (production volume) of engineered nano-TiO_2 was only 1.5 times larger than that of nano-FeOX . This is because in the former stud, a much higher percentage (60%) of engineered nano-TiO_2 is used in cosmetics, which is mainly released to waste water and finally discharged to surface water.

There are other modeling studies available for nano-iron (combining both nano-FeOX and NZVI in the same model) where the authors used the production amount (over 40,000 tons) and the product allocation (7 technological sectors) from one commercially market research report [39, 40, 130]. We also included the data from this report as one of the sources to define the production amount and as one of the methods to obtain the product allocation [63]. Both production amount and product distribution are associated with high uncertainty. By combining different sources with different DoB in our approach, we were able to obtain probability distributions and therefore had the possibility to quantify the uncertainty of the model results by providing probability ranges of flows and concentrations. Other differences between the Keller model [39, 40, 130] and ours are the selection of product categories and the values for the transfer factors, e.g. the removal efficiencies in WWTP and WIP. Another major difference is that the ‘recycling’ compartment was not considered in the Keller model. However, this is the compartment where about 40% of nano-FeOX from ‘PMC’ flows to in our model and thus represents an important driver of the flows.

The PEC values we provide cannot be validated at the moment because analytical measurements are not advanced enoguht to quantify trace amount of ENMs with a high background concentration of naturly occurring nanoparticles [33]. These authors stated that combing modelling with analytical method can provide an overall view of the presence of the nanomaterial in the environment [33]. This is especially important for iron oxides which are known to occur naturally in nano-scaled form in all environmental compartments [179]. The riverine suspended sediment concentration was reported up to 50 mg/L [180]. The share of particulate iron in suspended sediments in European rivers was investigated by Poulton and Raiswell (2002) and the average riverine particulate iron accounts for 1.4% (wt) [181]. Hence, the average concentration of particulate iron in European rivers can reach up to 70 µg/l. This result is three orders of magnitudes higher than the predicted concentration of nano-FeOX in this study, indicating that nano-FeOX particles only have an insignificant contribution to the total concentration of iron in the European environment.

The flows of nano-FeOX in technical compartments, such as waste materials and wastewater treatment plant effluents, can provide input data for environmental fate modeling studies. In addition, the predicted concentrations also can be a reference for policy makers to decide whether it is necessary to monitor an effluent or biosolid [182]

Our work also provides the first hazard assessment for nano-FeOX based on a dataset covering 11 species and 26 endpoint concentrations, but only for aquatic system due to data availability. The lower boundary of the PNEC (169 µg/l) is at least four orders of magnitude larger than the upper boundary of the PEC of nano-FeOX in surface water (36 ng/l in the EU). Therefore, the current expected amount of nano-FeOX does not has an adverse impact to organisms in surface waters and the potential risk posed by nano-FeOX in the surface water is predicted to be limited. There is a high variability associated with the various ecotoxicological endpoint concentrations that we have used to calculate the PNEC. The data cover the toxicity of different iron oxides (nano Fe_3O_4 and nano Fe_2O_3) but also of functionalized nano Fe_2O_3 particles. For instance, for *D. magna,* the smallest observed NOEC is 23 µg/l, which is about 200 times smaller than the largest value (5,000 µg/l). Here the lowest NOEC indicates the toxicity of bare Fe_3O_4 nanoparticles to *D. magna*. However, the other five sensitivity concentrations for *D. magna* correspond to Fe_3O_4 with four different coatings, indicating that the coating plays a critical role in determining the toxicity of nano iron-oxide-based materials [183]. Nano-FeOX is incorporated in products directly or in a modified form. However, there is no data available on production or use of functionalized nano-FeOX and it is therefore not possible to conduct a risk assessment only for modified iron oxides nanoparticles. In this case, to better estimate the hazards of nano-FeOX applied in products, sensitivity concentrations should cover both the non-modified and the modified form of nano-FeOX . The probabilistic approach used in this study is able to cope with this uncertainty when defining the PNEC value. A larger set of ecotoxicity endpoints of nano-FeOX in different forms under various test conditions would help to better determine the hazard of nano-FeOX .

In summary, using the current model framework plus the included applications and transfer factors, less than 15% of produced nano-FeOX was directly released to environment (Air, Soil, Water & sediment). Moverover, the results also indicate that, at the predicted release level, nano-FeOX does not endanger the aquatic organisms in the EU and Swiss systems.

A2.5 References

Anastasio C & Martin ST (2001) Atmospheric nanoparticles: Nanoparticles and the Environment (ed. by JF Banfield & A Navrotsky), pp. 293-349.

ANSES (2013) Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)

ANSES (2014) Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)

Arami H, Khandhar A, Liggitt D & Krishnan KM (2015) In vivo delivery, pharmacokinetics, biodistribution and toxicity of iron oxide nanoparticles. Chemical Society Reviews 44: 8576-8607.

Baumann J, Koser J, Arndt D & Filser J (2014) The coating makes the difference: acute effects of iron oxide nanoparticles on Daphnia magna. Sci Total Environ 484: 176-184.

Blazkova I, Nguyen HV, Dostalova S, Kopel P, Stanisavljevic M, Vaculovicova M, Stiborova M, Eckschlager T, Kizek R & Adam V (2013) Apoferritin Modified Magnetic Particles as Doxorubicin Carriers for Anticancer Drug Delivery. International Journal of Molecular Sciences 14: 13391-13402.

Bonnemain B (1998) Superparamagnetic agents in magnetic resonance imaging: Physicochemical characteristics and clinical applications - A review. J Drug Target 6: 167-174.

Caballero-Guzman A, Sun T & Nowack B (2015) Flows of engineered nanomaterials through the recycling process in Switzerland. Waste Manag 36: 33-43.

Cengelli F, Grzyb JA, Montoro A, Hofmann H, Hanessian S & Juillerat-Jeanneret L (2009) Surface-Functionalized Ultrasmall Superparamagnetic Nanoparticles as Magnetic Delivery Vectors for Camptothecin. Chemmedchem 4: 988-997.

Coll C, Notter D, Gottschalk F, Sun T, Som C & Nowack B (2016) Probabilistic environmental risk assessment of five nanomaterials (nano-TiO_2, nano-Ag, nano-ZnO, CNT, and fullerenes). Nanotoxicology 10: 436-444. doi:10.3109/17435390.2015.1073812.

Cornell RM & Schwertmann U (2003) The Iron Oxides: Structures, Properties, Reactions, Occurences and Uses. Wiley-VCH Verlag GmbH & Co. KGaA, Weinheim.

ECB (2003) Technical Guidance Document on Risk Assessment, European Chemicals Bureau.

ECHA (2008a) Guidance on information requirements and chemical safety assessment Part B: Hazard assessment.

ECHA (2008b) Guidance on information requirements and chemical safety assessment, Part E risk characterization.

EPR (2015) European Patent Office. European Patent Register Available from: http://www.epo.org/index.html. Retrieved 14.03.2015.

EUCommission (2011) Recommendation on the definition of nanomaterial (2011/696/EU). O. J. L 275: 3.

Future Markets (2012) The global market for nanomaterials 2002–2006: production volumes, revenues and end use markets. Future Markets Inc., http://www.futuremarketsinc.com/index.php?option=com_content&view=article&id= 176&Itemid=73.

Future Markets (2014) The Global Market for Metal Oxide Nanoparticles 2014. Future Market.

Garner KL, Suh S, Lenihan HS & Keller AA (2015) Species sensitivity distributions for engineered nanomaterials. Environ Sci Technol 49: 5753-5759.

Gottschalk F, Lassen C, Kjoelholt J, Christensen F & Nowack B (2015) Modeling flows and concentrations of nine engineered nanomaterials in the Danish environment. Int J Environ Res Public Health 12: 5581-5602.

Gottschalk F & Nowack B (2012) A probabilistic method for species sensitivity distributions taking into account the inherent uncertainty and variability of effects to estimate environmental risk. Integrated Environmental Assessment and Management 9: 79–86.

Gottschalk F, Scholz RW & Nowack B (2010) Probabilistic material flow modeling for assessing the environmental exposure to compounds: Methodology and an application to engineered nano-TiO2 particles. Environ. Modeling Software 25: 320-332.

Gottschalk F, Sonderer T, Scholz RW & Nowack B (2009) Modeled environmental concentrations of engineered nanomaterials (TiO_2, ZnO, Ag, CNT, fullerenes) for different regions. Environ. Sci. Technol. 43: 9216-9222.

H.B. Guo & Barnard AS (2013) Naturally occurring iron oxide nanoparticles: morphology, surface chemistry and environmental stability. Journal of Materials Chemistry A 1: 17.

Hazeem LJ, Waheed FA, Rashdan S, Bououdina M, Brunet L, Slomianny C, Boukherroub R & Elmeselmani WA (2015) Effect of magnetic iron oxide (Fe3O4) nanoparticles on the growth and photosynthetic pigment content of Picochlorum sp. Environmental Science and Pollution Research 22: 11728-11739.

Hincapie I, Caballero-Guzman A, Hiltbrunner D & Nowack B (2015) Use of engineered nanomaterials in the construction industry with specific emphasis on paints and their flows in construction and demolition waste in Switzerland. Waste Management 43: 398-406.

Hochella MFJ, Lower SK, Maurice PA, Penn RL, Sahai N, Sparks DL & Twining BS (2008) Nanominerals, mineral nanoparticles, and Earth systems. Science 319: 1631-1635.

K.Tockner, U.Uehlinger & C.T.Robinson (2009) Rivers of Europe.

Karlsson HL, Gustafsson J, Cronholm P & Möller L (2009) Size-dependent toxicity of metal oxide particles—A comparison between nano- and micrometer size. Toxicol Lett 188: 112-118.

Keller AA & Lazareva A (2014) Predicted Releases of Engineered Nanomaterials: From Global to Regional to Local. Environmental Science & Technology Letters 1: 65-70.

Keller AA, McFerran S, Lazareva A & Suh S (2013) Global life cycle releases of engineered nanomaterials. Journal of Nanoparticle Research 15: 17.

Konishi H, Xu Hf & Guo HB (2012) Nature's Nanostructures. Pan Stanford Pte. Ltd., Singapore.

Kunzmann A, Andersson B, Thurnherr T, Krug H, Scheynius A & Fadeel B (2011) Toxicology of engineered nanomaterials: Focus on biocompatibility, biodistribution and biodegradation. Biochimica Et Biophysica Acta-General Subjects 1810: 361-373.

Liu HH & Cohen Y (2014) Multimedia Environmental Distribution of Engineered Nanomaterials. Environmental Science & Technology 48: 3281-3292.

Luengo C, Brigante M, Antelo J & Avena M (2006) Kinetics of phosphate adsorption on goethite: Comparing batch adsorption and ATR-IR measurements. J Colloid Interface Sci 300: 511-518.

Mahapatra I, Sun TY, Clark JRA, Dobson PJ, Hungerbuehler K, Owen R, Nowack B & Lead J (2015) Probabilistic modelling of prospective environmental concentrations of gold nanoparticles from medical applications as a basis for risk assessment. J Nanobiotechnology 13: 1-14.

Mahmoudi M, Laurent S, Shokrgozar MA & Hosseinkhani M (2011) Toxicity Evaluations of Superparamagnetic Iron Oxide Nanoparticles: Cell "Vision" versus Physicochemical Properties of Nanoparticles. ACS Nano 5: 7263-7276. doi:10.1021/nn2021088.

Mahmoudi M, Simchi A, Imani M, Shokrgozar MA, Milani AS, Hafeli UO & Stroeve P (2010) A new approach for the in vitro identification of the cytotoxicity of superparamagnetic iron oxide nanoparticles. Colloids Surf B Biointerfaces 75: 300-309.

Mahmoudi M, Simchi A, Milani AS & Stroeve P (2009) Cell toxicity of superparamagnetic iron oxide nanoparticles. J Colloid Interface Sci 336: 510-518.

Mikkel Aaman Sørensen, Fleming Ingerslev, Katrine Bom, Carsten Lassen, Frans Christensen & Marlies Warming (2015) Survey of products with nanosized pigments.

Miyata T, Ishino Y & Hirashima T (1978) Catalytic Reduction of Aromatic Nitro-Compounds with Hydrazine Hydrate in Presence of Iron(Iii) Oxide Hydroxide. Georg Thieme Verlag 1978: 834-835.

Mohapatra M & Anand S (2010) Synthesis and applications of nano-structured iron oxides/hydroxides- a review. International Journal of Engineering, Science and technology 2: 20

Mueller NC, Braun J, Bruns J, Černík M, Rissing P, Rickerby D & Nowack B (2012) Application of nano zero valent iron (nZVI) for groundwater remediation in Europe. Environ. Sci. Pollut. Res. 19: 550-558.

Mueller NC & Nowack B (2008) Exposure modeling of engineered nanoparticles in the environment. Environ Sci Technol 42: 4447-4453.

Navrotsky A, Mazeina L & Majzlan J (2008) Size-driven structural and thermodynamic complexity in iron oxides. Geochimica Et Cosmochimica Acta 72: A673-A673.

Nowack B, Baalousha M, Bornhoft N, Chaudhry Q, Cornelis G, Cotterill J, Gondikas A, Hassellov M, Lead J, Mitrano DM, von der Kammer F & Wontner-Smith T (2015) Progress towards the validation of modeled environmental concentrations of engineered nanomaterials by analytical measurements. Environmental Science-Nano 2: 421-428.

Pereira MC, Oliveira LCA & Murad E (2012) Iron oxide catalysts: Fenton and Fenton-like reactions - a review. Clay Minerals 47: 285-302.

Poulton SW & Raiswell R (2002) The low-temperature geochemical cycle of iron: From continental fluxes to marine sediment deposition. American Journal of Science 302: 774-805. doi:DOI 10.2475/ajs.302.9.774.

Praetorius A, Arvidsson R, Molander S & Scheringer M (2013) Facing complexity through informed simplifications: a research agenda for aquatic exposure assessment of nanoparticles. Environmental Science: Processes & Impacts 15: 161-168. doi:10.1039/C2EM30677H.

Prodan AM, Iconaru SL, Ciobanu CS, Chifiriuc MC, Stoicea M & Predoi D (2013) Iron Oxide Magnetic Nanoparticles: Characterization and Toxicity Evaluation by In Vitro and In Vivo Assays. Journal of Nanomaterials 2013: 1-10. doi:10.1155/2013/587021.

Rajabi F, Kakeshpour T & Saidi MR (2013) Supported iron oxide nanoparticles: Recoverable and efficient catalyst for oxidative S-S coupling of thiols to disulfides. Catalysis Communications 40: 13-17.

Semenzin E, Lanzellotto E, Hristozov D, Critto A, Zabeo A, Giubilato E & Marcomini A (2015) Species sensitivity weighted distribution for ecological risk assessment of engineered nanomaterials: the n-TiO2 case study. Environmental Toxicology and Chemistry 34: 2644-2659.

Sposito G (1989) Iron oxides: The chemistry of soils (ed. Oxford University press, , New York, p. 277.

Sreeram KJ, Indumathy R, Rajaram A, Nair BU & Ramasami T (2006) Template synthesis of highly crystalline and monodisperse iron oxide pigments of nanosize. Materials Research Bulletin 41: 1875-1881.

SRIConsulting (2011) The Chemical Economics Handbook: Inorganic chemical pigments: Inorganic color pigments (ed., pp. 8-48.

Sumner ME (1963) Effect of iron oxides on positive and negative charges in clays and soils. Clay Minerals: 218-226.

Sun TY, Bornhoft NA, Hungerbuhler K & Nowack B (2016) Dynamic Probabilistic Modeling of Environmental Emissions of Engineered Nanomaterials. Environ Sci Technol 50: 4701-4711.

Sun TY, Gottschalk F, Hungerbühler K & Nowack B (2014) Comprehensive Modeling of Environmental Emissions of Engineered Nanomaterials. Environ. Pollut. 185: 69-76.

Tran N & Webster TJ (2010) Magnetic nanoparticles: biomedical applications and challenges. Journal of Materials Chemistry 20: 8760-8767.

Uheida A, Iglesias M, Fontas C, Hidalgo M, Salvado V, Zhang Y & Muhammed M (2006a) Sorption of palladium(II), rhodium(III), and platinum(IV) on Fe3O4 nanoparticles. J Colloid Interface Sci 301: 402-408.

Uheida A, Salazar-Alvarez G, Bjorkman E, Yu Z & Muhammed M (2006b) Fe3O4 and gamma-Fe2O3 nanoparticles for the adsorption of Co2+ from aqueous solution. J Colloid Interface Sci 298: 501-507.

USP&TO (2015) United States Patent and Trademark Office. Patent Full-Text Databases Available from: http://patft.uspto.gov/netahtml/PTO/index.html. Retrieved 14.03.2015.

Walser T, Limbach LK, Brogioli R, Erismann E, Flamigni L, Hattendorf B, Juchli M, Krumeich F, Ludwig C, Prikopsky K, Rossier M, Saner D, Sigg A, Hellweg S, Gunther D & Stark WJ (2012) Persistence of engineered nanoparticles in a municipal solid-waste incineration plant. Nat Nano 7: 520-524.

Wang Y, Kalinina A, Sun T & Nowack B (2016a) Probabilistic modeling of the flows and environmental risks of nano-silica. Sci Total Environ 545-546: 67-76.

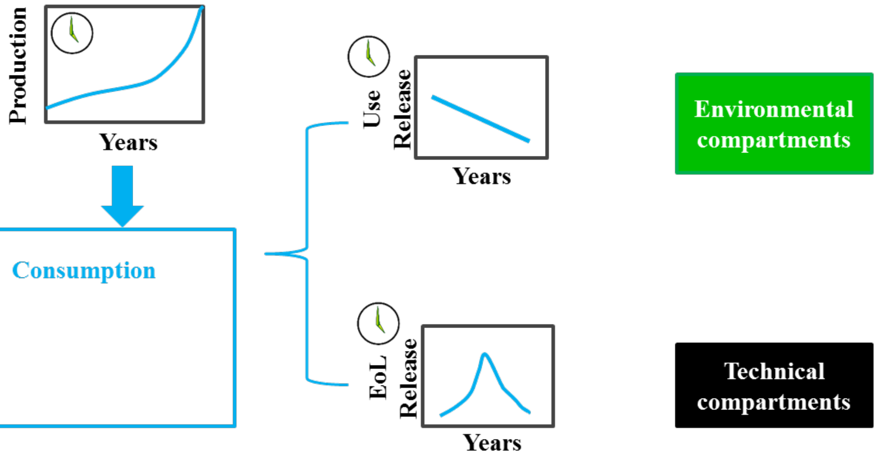
Wang Y, Kalinina A, Sun T & Nowack B (2016b) Probabilistic modeling of the flows and environmental risks of nano-silica. Science of the Total Environment 545-546: 67-76.

Weissleder R, Stark DD, Engelstad BL, Bacon BR, Compton CC, White DL, Jacobs P & Lewis J (1989) Superparamagnetic Iron-Oxide - Pharmacokinetics and Toxicity. American Journal of Roentgenology 152: 167-173.

Zhang Y, Zhu L, Zhou Y & Chen JM (2015) Accumulation and elimination of iron oxide nanomaterials in zebrafish (Danio rerio) upon chronic aqueous exposure. Journal of Environmental Sciences-China 30: 223-230.

Zhu XS, Tian SY & Cai ZH (2012) Toxicity Assessment of Iron Oxide Nanoparticles in Zebrafish (Danio rerio) Early Life Stages. Plos One 7: e46286.

Annex 3. Dynamic probabilistic material flow analysis of nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots in seven European regions



Submitted to Environmental Pollution

Abstract

Static environmental exposure assessment models based on material flow analysis (MFA) have previously been used to estimate flows of engineered nanomaterials (ENMs) to the environment. However, such models do not account for changes in the system behavior over time. Dynamic MFA used in this study includes the time-dependent development of the modelling system by considering accumulation of ENMs in stocks and the environment, and the dynamic release of ENMs from nano-products. In addition, this study also included regional variations in population, waste management systems, and environmental compartments, which subsequently influence the environmental release and concentrations of ENMs. We have estimated the flows and release concentrations of nano-SiO_2, nano-iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots in the EU and six geographical sub-regions in Europe (Central Europe, Northern Europe, Southern Europe, Eastern Europe, South-eastern Europe, and Switzerland). The model predicts that a large amount of ENMs are accumulated in stocks (not considering further transformation). For example, in the EU 2,040 Mt of nano-SiO_2 are stored in the in-use stock, 80,400 tonnes have been accumulated in sediments and 65,600 tonnes in natural and urban soil from 1990 to 2014. The magnitude of flows in waste management processes in different regions varies because of differences in waste handling. For example, concentrations in landfilled waste are lowest in South-eastern Europe due to dilution by the high amount of landfilled waste in the region. The flows predicted in this work can serve as improved input data for mechanistic environmental fate models and risk assessment studies compared to previous estimates using static models.

A3.1 Introduction

Due to their unique properties, nanomaterials are being increasingly used in a wide range of consumer products and industrial applications and a sizeable nano-industry has rapidly developed. This, in turn, has led to a rise in the release of engineered nanomaterials (ENMs) to the environment, which is a concern due to their potentially negative effects on organisms (Chen et al. 2011, Klaine et al. 2012, Money et al. 2012). However, evaluating the environmental impacts of ENMs is challenging due to difficulties in quantifying ENM concentrations in the environment by analytical measurements (Nowack et al. 2015). Given these challenges, material flow analysis (MFA) models and environmental fate models have emerged as the best alternative approach for estimating ENM release amounts (Baalousha et al. 2016, Dale et al. 2015a, Hendren et al. 2013, Nowack 2017).

Mueller and Nowack (2008) developed the first generation of a nano-MFA model based on a life cycle concept and applied it to estimate concentrations of nano-TiO_2, nano silver, and carbon nanotubes (CNT) in air, soil, and water. Later, Gottschalk et al., (2010) proposed a second generation model based on a probabilistic approach, with uncertain parameters described by probabilistic distributions. This model has been applied to estimate the environmental flows and concentrations of different ENMs in Europe, the USA and Switzerland (Gottschalk et al. 2009, Mahapatra et al. 2015, Sun et al. 2014, Wang et al. 2016a, Wang et al. 2016b). However, these models have two important limitations: a) the input is only for a single, specific year, which might result underestimating flows to and concentrations in the environment, especially for compartments, where ENMs are likely accumulated, and b) all ENMs are assumed to be released during the same year in which they are produced. This may overestimate the amount of ENMs in the compartments, where ENMs run through, e.g. surface water. Hence, by not considering time-dependent input and release, these models simplify the behaviour of complex, dynamic systems.

To address these problems, a third generation model was developed by Bornhöft et al (2016), the dynamic probabilistic MFA (DPMFA). This model considers the historic input of ENMs from previous time periods and the residence time of ENMs in nano-products. With the DPMFA model, it is possible to calculate the amount of ENMs stored in products in the use phase (henceforth, *in-use stock*) from where they will likely be released or disposed along nano-products in the future, as well as the amounts of ENMs that are stored in environmental compartments and waste management processes (*sinks*). DPMFA has been applied by Sun et al (2016) to estimate the emissions of nano-TiO_2, nano-ZnO, nano-Ag and CNT to the environment and the environmental stocks in the EU from 1990 to 2020. The authors suggest that a significant share of ENMs remain in-use stock for a period of time instead of being released immediately in the year of their production. Due to the consideration of accumulation of ENMs in in-use stocks and the environment, the DPMFA provides a more rational means of predicting ENM concentrations in the environment compared to traditional static MFA models. The model has also been applied to predict future releases under different scenarios (Sun et al. 2017).

MFA models have been applied to estimate releases and environmental concentrations of various ENMs, covering nano-CeO_2, nano-Al_2O_3, fullerenes, nano-gold, nano-Ag, nano-SiO_2, nano-TiO_2, nano-ZnO, CNTs, and quantum dots (QD) (Boxall et al. 2007, Gottschalk et al. 2015, Gottschalk et al. 2009, Keller &Lazareva 2014, Keller et al. 2013, Keller et al. 2014, Mueller &Nowack 2008, O'Brien &Cummins 2008, Sun et al. 2014, Wang et al. 2016a, Wang et al. 2016b). Studies have also applied MFA to nano-$CuCO_3$, carbon black and nano-clays (Gottschalk et al. 2015, Keller &Lazareva 2014). Among the studied ENMs, nano-TiO_2, nano-ZnO, nano-silver, CNTs, and fullerenes are the most intensively studied materials using various modelling approaches. Only few modelling studies have covered nano-SiO_2, nano-iron oxides, nano-CeO_2 and nano-Al_2O_3 (Boxall et al. 2007, Keller &Lazareva 2014, Keller et al. 2013, Wang et al. 2016a, Wang et al. 2016b), and only one included so far QDs (Gottschalk et al. 2015). Given the increasing use and importance of these materials in commercial products (Future Markets 2014), it is imperative to estimate their environmental release using updated data on production and use.

Previous modelling studies have predicted environmental exposure of ENMs at global, regional and local scales (Dumont et al. 2015, Gottschalk et al. 2011, Johnson et al. 2011, Keller &Lazareva 2014, Keller et al. 2013, Sun et al. 2015). Gottschalk et al (2011) investigated the exposure of ENMs in Swiss surface water with a high temporal and spatial resolution and found that local concentrations varied by up to a factor of 10 between different locations. Keller et al (2014) have predicted the worldwide release level of 10 different ENMs. Later they conducted another MFA study that considers differences in consumption levels and waste management strategies to estimate exposure of the same ENMs in different regions, e.g. Europe, Asia, North America. They predicted that Asia has the highest consumption and release of ENMs, followed by Europe and North America. Another recent modelling study by Dumont et al (2015) has estimated the concentration of nano-Ag and nano-ZnO in European surface waters at high spatial and temporal resolution, and higher concentrations were typically found in water systems near large cities. The surface water concentrations in Eastern and Southern Europe were modelled to be three orders of magnitude higher than in Northern Europe. It is clear that economic and environmental differences between sub-regions of the EU affect the mass flows of ENMs through these separate systems. For instance, previous MFA studies used average values of transfer factors to describe the mass transport of ENMs between waste management processes and environmental compartments, to represent the EU. Specifically, the share of wastewater treated in sewage treatment plants in the EU was considered to be on average 80% in previous MFA studies (Sun et al. 2016, Wang et al. 2016b). However, the connection rate in Central Europe is much higher than in South-eastern Europe (EEA. 2013), which results in less release of ENMs directly to surface water in Central Europe compared to Southern Europe. To ensure that these potentially important differences between sub-regions of Europe are duly considered, it is important that modelling is undertaken at least at a regional scale.

There is still a lack of information regarding the environmental exposure level of the engineered nano-SiO_2, nano-iron oxides, nano-CeO_2, nano-Al_2O_3, and QDs. The aim of this study is thus to predict the flows of nano-SiO_2, nano-iron oxides, nano-CeO_2, nano-Al_2O_3, and QDs to the environment using a dynamic model from 1990 to 2020 in seven geographic systems: the EU as a whole, Central Europe, Northern Europe, Southern Europe, Eastern Europe, South-eastern Europe, and Switzerland.

A3.2 Methods

A3.2.1 Exposure model

A3.2.1.1 Model framework

The model used in this paper is based on the dynamic probabilistic material flow (DPMFA) model developed by Bornhöft et al (2016) and previously applied by Sun et al (2016) and Sun et al. (2017) to predict flows and concentrations of ENMs in the EU. The model is dynamic in respect to its treatment of input flows and releases of nanomaterials into the environment. Whereas previous static MFA models used ENM production volume data for a single, specific year as input, the DPMFA considers the increase of ENM production over multiple years. Dynamic release into the environment is reflected by the time-dependent release of ENMs from nano-products. ENMs may be released to the environment during any stage of the life cycle, i.e. during production, manufacturing, and use (Koivisto et al. 2017, Wohlleben &Neubauer 2016), which cause environmental impact through releases during every life cycle stage (Zhang et al. 2016).The dynamic model considers the whole product life cycle from the production of the ENMs, the manufacturing of nano-products, their use/application, and, finally, their end-of-life (EoL) management. Figure 1 shows the structure of the DPMFA model. Following production, ENMs are incorporated into nano-products and subsequently used. The release of ENMs from nano-products may occur during the use phase (use release) or during the EoL phase (EoL release). Therefore, the total release in a specific year consists of use release and EoL release. Furthermore, the releases of ENMs may occur during the same year in which they were produced (immediate release) or in subsequent years (delayed release). These releases are distributed amongst environmental compartments and waste management processes.

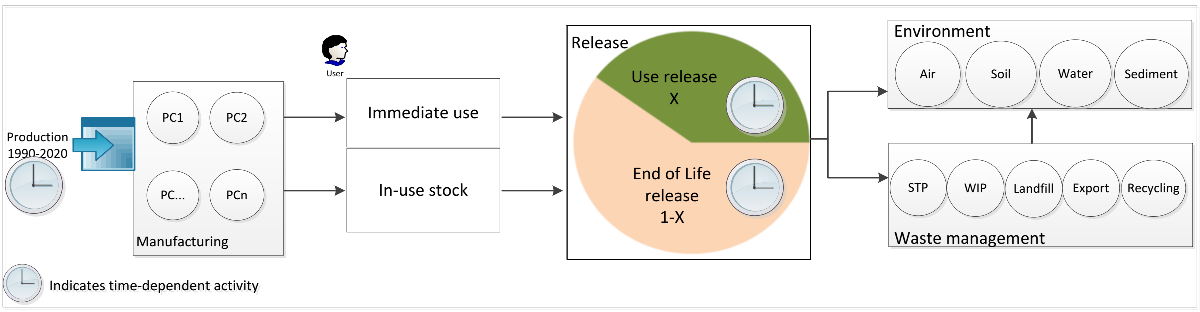


Figure A3.1. The structure of the dynamic probabilistic material flow analysis (DPMFA) model. In the model, ENMs produced in a given year are allocated to different product categories (PC). For each annual time step, ENMs are either released immediately (immediate use) or stored in the in-use stock and are released in later periods of time (delayed release). The release box consists of releases from the use phase (use release), and from the end of life stage (end of life release). The environmental compartments are air, soil, water and sediment. The considered waste management processes are STP, waste incineration plants (WIP), landfills, export and recycling.

The required input parameters for the model are the annual production volumes of ENMs for the period 1990-2020, the share of ENMs in defined product categories, the release schedules of ENMs from different product categories and transfer factors between waste management processes and environmental compartments. Products can be treated or disposed of via a number of different waste management processes. The waste management processes considered here are landfilling, waste incineration plants, sludge treatment plants, recycling, and export. Export represents a process through which ENMs flow out of the system boundary, e.g. some material might be recycled in other geographical regions due to economical reasons (Caballero-Guzman &Nowack 2016). The environmental compartments are air, natural and urban soil, sludge treated soil, surface water, and sediment. Landfill, soil, and sediment were considered as final sinks.

The time period considered in this study were the years 1990-2020. Data on the estimated development of nanotechnology from 1990 to 2020 were taken from Sun et al (2016). Data on the production volumes of an ENM for a specific year, if not available, was extrapolated from available data for other years based on the estimates in Sun et al (2016).

A3.2.1.2 Assumptions

The environmental fate of ENMs was not included in the model. It was assumed that no leachate from landfills or run-off from soil reaches surface waters. At the end of the life-cycle, products may be treated in different recycling methods after flowing to the recycling system. The outflows of ENMs from recycling were only considered for those product categories where data were available (Caballero-Guzman et al. 2015). Based on the literature, the residence time of ENM in air was assumed to be the same than that of ultrafine particles (10 days, Anastasio &Martin 2001). The residence time of water was set to 40 days based on ECHA 2016.

Data from different sources were often inconsistent, for example, the production volume of target ENM varies a lot from one source to another, and data on the release amount of ENMs from their applications are largely missing. In many cases data were not available at all, e.g. the release amount of ENMs from their applications. In these cases, experts were consulted to overcome such data gaps. To account for the uncertainty in parameter values that stems from our incomplete knowledge, Bayesian distributions were applied to each parameter value, with outputs (e.g. release and concentration of ENMs in soil) calculated through Monte Carlo simulations (10,000 iterations) (Bornhoft et al. 2016). With this approach, DPMFA is able to translate the uncertain input parameters to the model and express the uncertainty in each output variable based on 15% and 85% quantiles.

A3.2.2 Geographical regionalization

31 countries were considered: the 27 countries of the EU plus Iceland, Norway, Switzerland, and Turkey. These countries were grouped into seven regions: the EU 27 (EU), Switzerland (CH), Central Europe (CE), Eastern Europe (EE), Southern Europe (SE), northern Europe (NE), and South-eastern Europe (SEE). Details of the grouping of the countries are shown in Table S1. The grouping of countries was based on a report on urban wastewater treatment by the European Environmental Agency (EEA. 2013).

Geographic differences in waste management processes were taken into account, including (a) wastewater connection rate to sewage treatment plants (the remainder is discharged directly to surface waters); (b) biosolids use on agricultural land; and (c) municipal waste and construction waste management to determin how much ENMs flow through EoL products to landfill, waste incineration plant, recycling, and export. The detailed information regarding the waste management for each EoL product category is presented in Table 1 and Table S2. If data is missing for EoL products in any region, the number for the general EU was chosen for the region.

A3.2.3 Input parameters and their allocation

A3.2.3.1 Production volume

Data were collected from various sources, e.g. scientific literature, market reports, and expert consultation. Since the production volumes of ENMs in a given year vary considerably between sources, a weighting factor was given to each data source based on their relative degrees of uncertainty. Figure S1 in the SI illustrates the decision tree to assign a weight to each source. The collected production data are shown in Table S2 in the SI. The production value for each target region was allocated from the global production amount using the GDP (PPP) (Gross Domestic Product per capita based on Purchasing Power Parity) (Wang et al. 2016a, Wang et al. 2016b), as well as the market demand of each region (Future Markets 2014). A weight was also assigned to each scaling factor (shown in Table S2). In this paper, we assumed that the amount of ENMs used in each region equals to their production volume.

A3.2.3.2 Application of ENMs

Information related to types of applications in which ENMs are used was obtained from several databases. Table S4 shows the approaches used to estimate the share of ENM use between product categories, as well as details of the method for weighting data sources (see Figure S2). The allocation of the five ENMs to the different product categories is shown in Figure 2. Significant percentages of nano-SiO_2 and nano-iron oxides are used in paints, 42% for nano-SiO_2 and 54% for nano-iron oxides, respectively. More than 50% of nano-Al_2O_3 is relatively evenly distributed among the categories ’Filter’, ‘Paint’, ‘Cement’, and ‘Cosmetics & personal care’. ‘Polishing’ (42%) ‘Fuel additive’ (16%), and ‘Glass’ (13%) are the dominant categories for nano-CeO_2. The two main uses of QDs are in medical applications and electric devices. The shares of nano-SiO_2 and nano-iron oxides in each product category in this work are different from those in the previous static MFA studies, because an additional method based on the market size is included in the model to obtain the product distribution (see SI for details) (Wang et al. 2016a, Wang et al. 2016b). It was assumed that there are no temporal or spatial differences in the shares of ENMs between product categories in the current study.

A3.2.3.3 Release schedule and distribution

To simulate the time-dependent nature of ENM releases, the following aspects were considered: the life time of each product category, the annual release rate during the use phase and the distribution of these releases between environmental compartments and finally the release during the EoL phase and the distribution of these releases between waste management processes. Wohlleben and Neubauer (2016) have shown that the release of ENMs from nanoproducts is mostly determined by the matrix and only marginally by the type of ENM (Wohlleben &Neubauer 2016). Therefore, release data for the product categories in Sun et al (2016) were used in this work. Data regarding wastewater treatment, solid waste handling, and lifespans of products were also taken from (European Commission 2010, Eurostat 2014, JNIES 2016) and from waste expert at Empa . Product lifespans were assumed to be consistent with those reported in the *Lifespan Database for Vehicles, Equipment, and Structures* published by the Japanese National Institute for Environmental Studies (JNIES 2016). If data were not available from this database, they were taken from the literature or estimated. As an example of a release schedule, the values for QDs are presented in Table 1. The same data for the other ENMs are given in Table S5.

The waste management processes in each region were assumed to be consistent with the average situation in the EU for the case of QD. However, for product categories containing other ENMs, e.g. 'Plastic', the regional differences in waste management processes was included in the model (see Table S5). The distribution of the considered ENMs among waste management processes depends on the products within which they are contained. QDs are commonly incorporated into the matrix of electrical products, which are sent for waste management at the end of their lives and are predominantly transferred to recycling, as shown in Table 1. Only a small amount of QDs are released during the use phase (from product categories "Catalyst" and "Ink") to wastewater treatment and air.

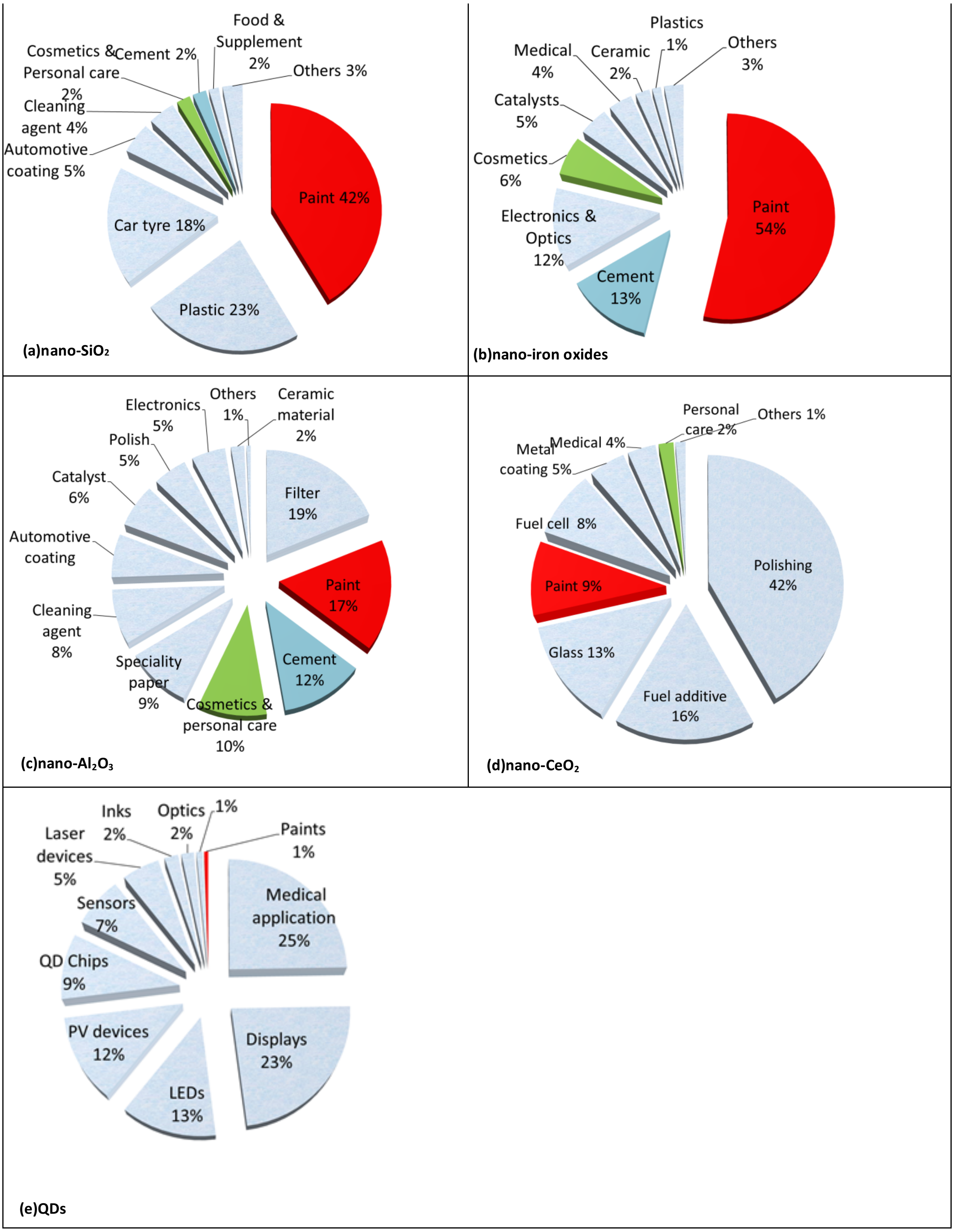


Figure A3.2. The allocation of the ENMs considered in this work to different product categories (a) nano-SiO_2, (b) nano-iron oxides, (c) nano-Al_2O_3, (d) nano-CeO_2, (e) QDs. The common categories of ENMs are in the same color. ‘Paint’ in red; ‘Cosmetics & personal care’ in green; ‘Cement’ in blue.

A3.2.4 Predicted Environmental Concentrations

Concentrations of ENMs in waste management processes and environmental compartments were calculated by dividing the released mass of ENMs in each process and compartment by the mass or volume of the compartment, as shown in Table A3.6. It was assumed that all compartments are homogenous and well-mixed [200].

*Table A3.1. Release schedule of Quantum dots (QD). The table gives information regarding QDs release during use (in light green) and at the EoL stages (in light grey). The use release part describes how much QDs are released during the use phase ("use release amount"), how long this release lasts ("use release duration"), how much is released in each year ("use release schedule"), and the compartment to which the QDs are flowing ("use release distribution"). For the EoL release (grey cells), the same approach is used. Lifetimes are modelled using normal distributions. Y1 = Year 1, Y2 = Year 2. WIP: plant. * are based on expert judgments.*

QD	**Use Release amount**	**Use Release duration**	**Use release schedule**		**Use release distribution**				**EoL Release amount**	**Lifetime distribution parameters (years)**	**EoL release distribution**				**Reference**
Product Category	**x**		**Y1**	**Y2**	**Waste water**	**Air**	**Surface water**	**Soil**	**1-x**		**Landfill**	**WIP**	**Recycling**	**Export**	
Catalysts	0.01 *	1	1		1				0.99	1 *			1*		
QD chips	0 [190]								1	mean=10, 3σ= 5 [203]	0.09	0.06	0.65	0.2	[190]
Displays	0 [190]								1	mean=10, 3σ= 5 [203]	0.09	0.06	0.65	0.2	[190]
Inks	0.005 *	1	1			1			0.995	1	0.07	0.03	0.7	0.2	[190]
Laser devices	0 [190]								1	mean=10, 3σ= 5 [203]			1*		
LEDs	0 [190]								1	mean=14, 3σ=5 [203]	0.09	0.06	0.65	0.2	[190]
Medical application	0 [190]								1	Y=0.9, Y=0.1 *		1*			
Optics	0 *								1	mean=40, 3σ=10 *		1*			
Paints	0 *								1	mean=25, 3σ=10 [203]	0.05*	0.05*	0.9*		
PV devices	0 *								1	mean=30, 3σ=10 [190]			1*		
Sensors	0 *								1	mean=8, 3σ=8 [190]	0.09	0.06	0.65	0.2	[190]

A3.3 Results

A3.3.1 Probabilistic dynamic modelling

The main output of the model are probability distributions of masses of the five ENM for each year from 1990 to 2020. Figure 3 shows the development of the masses of nano-CeO_2 in different compartments over the considered 31-year time period: the use amount (a), the mass in the in-use stock (b), the mass accumulated in soils (c) and in sediments (d). Every grey dot corresponds to an iteration from the Monte Carlo simulation. The number of dots in Figure 3 is limited to 2000 to enhance visualization (10000 iterations in total). Mean values (red lines) are given in each subplot, as well as the 15% and 85% quantiles (green dashed lines). Figure 3a shows that the production volume of nano-CeO_2 has increased, and continues to increase substantially, particularly since 2013. This rapidly increasing rate of production, unsurprisingly, induces a substantial growth of nano-CeO_2 in the in-use stock as well as in the environmental sink compartments (soil and sediment). Comparing Figure 3b with Figure 3a, it is visible that the amount of nano-CeO_2 stored in the in-use stock is larger than the annual production volume, with 3,240 tonnes of nano-CeO_2 in the in-use stock in 2014 while 1,300 tonnes were produced in the same year. Moreover, Figure 3c and 3d demonstrate that a significant amount of nano-CeO_2 has already accumulated in soils and sediments, with the amount in soils being twice larger than that in sediments. This is because nano-CeO_2 flows to soil from deposition from air, and from sludge application.

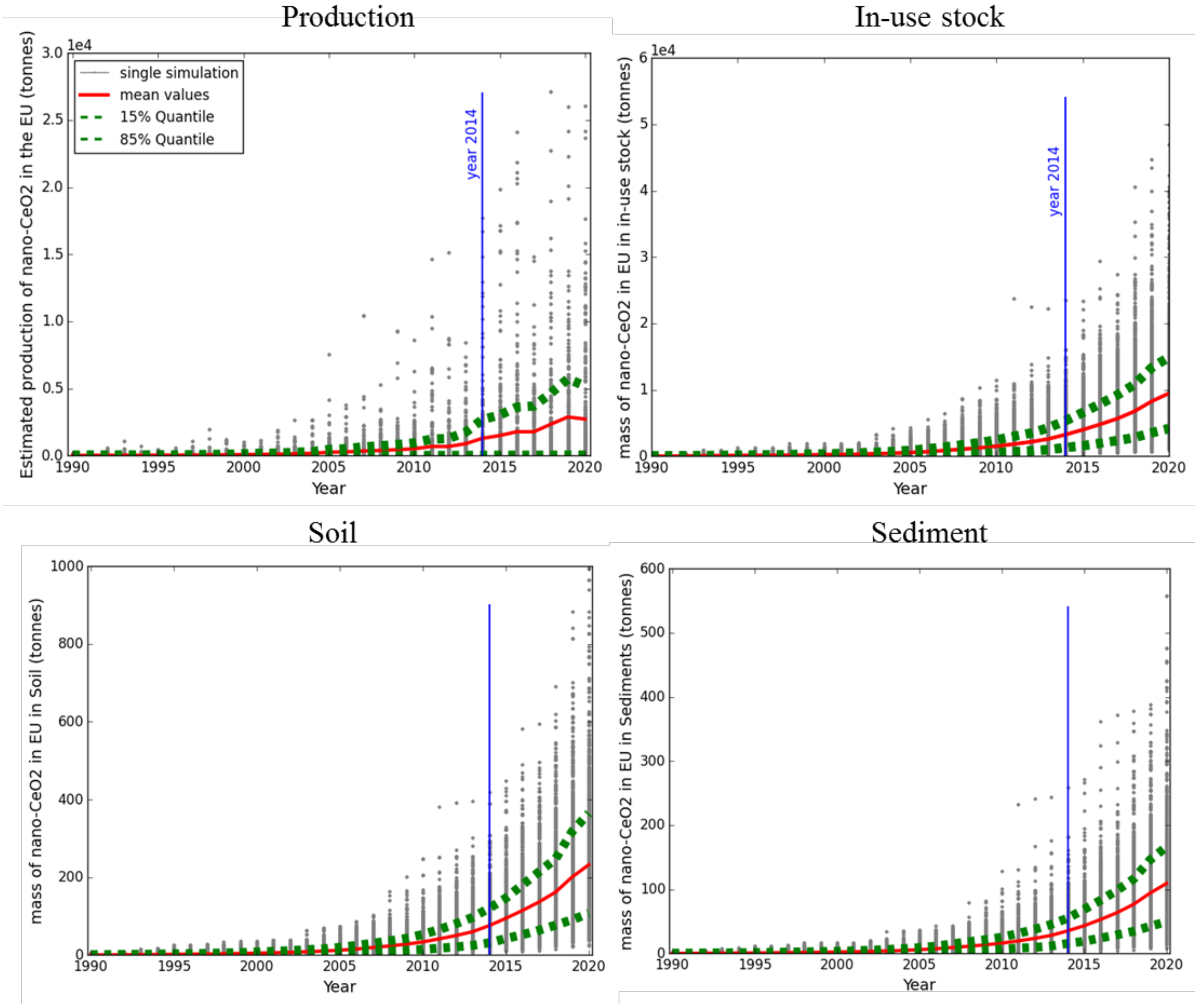


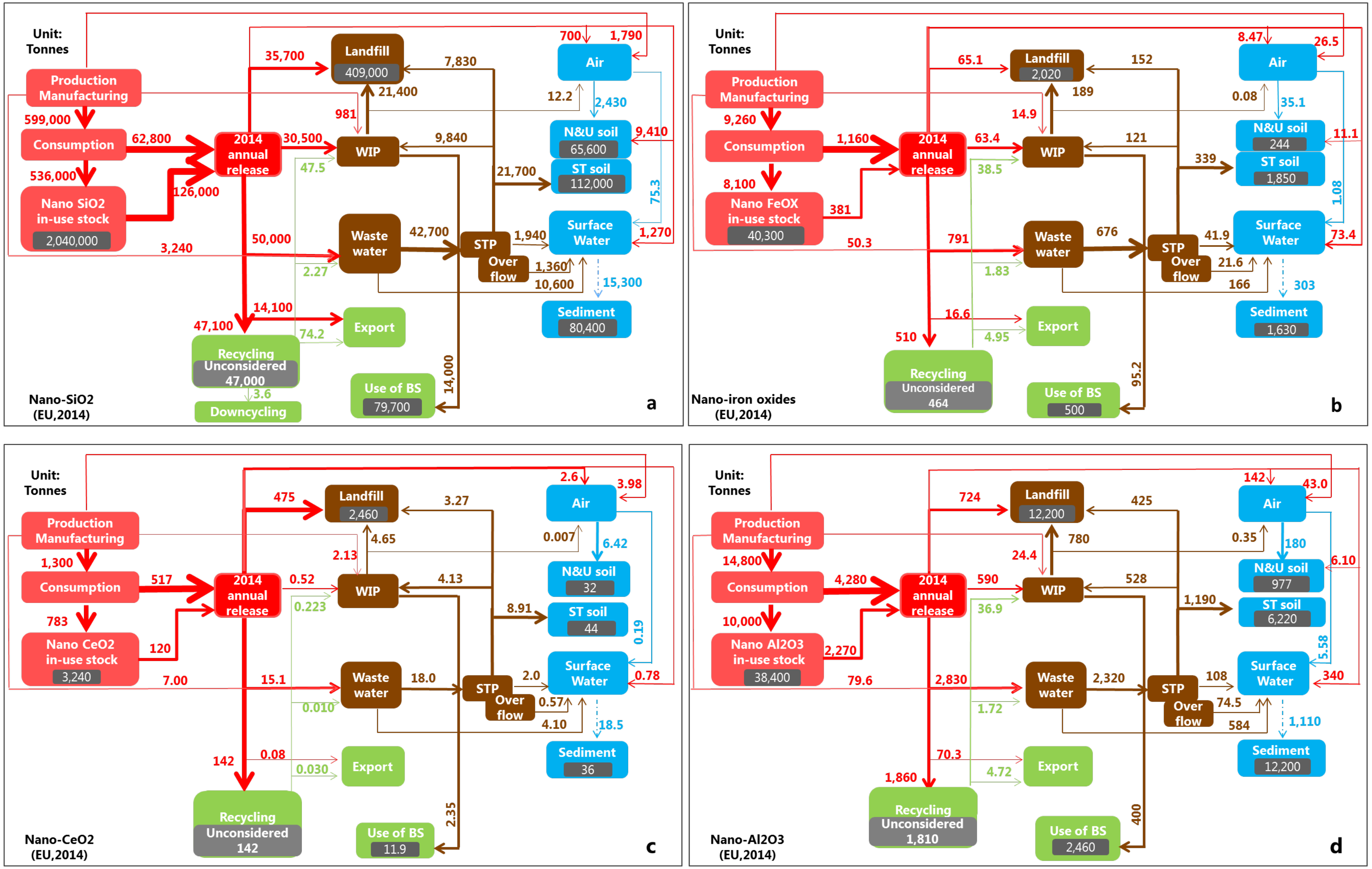
Figure A3.3. Evolution of nano-CeO_2 in the EU from 1990 to 2020 (tonnes) in (a) production, (b) in-use stock, (c) soil, and (d) sediment. The grey dots represent a single simulation. The two green dashed lines in each subplot are the 15% and 85% quantiles of the probability distributions of each year. The red lines are the mean values. The blue vertical lines indicate the reference year (2014).

A3.3.2 Mass flows

As 2014 was the reference year in our static modeling studies for nano-SiO_2 and nano-iron oxides, the mass flows of the ENMs considered in this work were also calculated for 2014. To estimate the release of ENMs in 2014, we considered 25 years of input from 1990 to 2014, combining direct and delayed release ENMs from nano-products. As shown in Figure 4, the ENM flows from 'Production & Manufacturing' to 'Consumption' are split into the in-use stock (where ENMs are temporarily stored) and immediate use (corresponding to immediate release). Between the five ENMs considered, the greatest estimated amount consumed in the EU in 2014 is observed for nano-SiO_2 (599,000 t), followed by nano-Al_2O_3 (14,800 t), nano-iron oxides (9,260 t), nano-CeO_2 (1,300 t), and QDs (413 kg). The estimated use volume of nano-SiO_2 in the EU is more than six times higher than in our previous study (Wang et al. 2016b). This can be explained by the data from additional sources, e.g. Future Market (2014), or SRI consulting (2011) (see Table S2), that are several times higher than the mean value in the study published previously (Future Markets 2014, SRIConsulting 2011).

For all considered ENMs, more than 60% of them flow to the in-use stock, which results a delayed release. Many ENMs are typically incorporated into products that have lifetimes longer than one year. Two thirds of the total releases of nano-SiO_2 in 2014 are from the in-use stock, having entered the use stage in previous years, whereas for nano-CeO_2 this number is less than 20%. This is because 42% of nano-CeO_2 is used in chemical and mechanical polishing, an application which results in immediate release. For the other three ENMs, the delayed release accounts for 25-35% of the total release. The differences are, again, due to the variations in applications between ENMs and the life times of the nano-products.

Using the DPMFA model, we are able to calculate the quantity of ENMs that are permanently or temporarily stored in environmental compartments. The amount of ENMs in the in-use stock is over two to five times larger than the annual input in 2014. The gap is even greater when the flow to sink compartments is compared with the total amount of ENMs in those compartments (landfill, natural and urban soil, sludge treated soil, and sediments). For example, the total stock of nano-SiO_2 in landfills in 2014 is about ten times that of the annual landfill input. Therefore, considering the accumulation of ENMs over time allow us to better estimate the total amount of ENMs in environmental compartment.


Unit: Tonnes
Production Manufacturing
599,000
Consumption
62,800
536,000
Nano SiO2 in-use stock
2,040,000
126,000
3,240
2014 annual release
35,700
30,500
Landfill
409,000
21,400
981
WIP
9,840
7,830
47.5
50,000
Waste water
42,700
2.27
STP
Over flow
1,940
1,360
10,600
21,700
14,100
47,100
74.2
Export
Recycling
Unconsidered
47,000
3.6
Downcycling
Use of BS
79,700
14,000
Air
700
1,790
12.2
2,430
N&U soil
65,600
9,410
ST soil
112,000
75.3
Surface Water
1,270
15,300
Sediment
80,400
Nano-SiO2 (EU,2014)
a
Unit: Tonnes
Production Manufacturing
9,260
Consumption
1,160
8,100
Nano FeOX in-use stock
40,300
381
50.3
2014 annual release
65.1
63.4
Landfill
2,020
189
14.9
WIP
121
152
38.5
791
Waste water
676
1.83
STP
Over flow
41.9
21.6
166
339
16.6
510
4.95
Export
Recycling
Unconsidered
464
Use of BS
500
95.2
Air
8.47
26.5
0.08
35.1
N&U soil
244
11.1
ST soil
1,850
1.08
Surface Water
73.4
303
Sediment
1,630
Nano-iron oxides (EU,2014)
b
Unit: Tonnes
Production Manufacturing
1,300
Consumption
517
783
Nano CeO2 in-use stock
3,240
120
7.00
2014 annual release
475
0.52
Landfill
2,460
4.65
2.13
WIP
4.13
3.27
0.223
15.1
Waste water
18.0
0.010
STP
Over flow
2.0
0.57
4.10
8.91
142
0.08
0.030
Export
Recycling
Unconsidered
142
Use of BS
11.9
2.35
Air
2.6
3.98
0.007
6.42
N&U soil
32
ST soil
44
0.19
Surface Water
0.78
18.5
Sediment
36
Nano-CeO2 (EU,2014)
c
Unit: Tonnes
Production Manufacturing
14,800
Consumption
4,280
10,000
Nano Al2O3 in-use stock
38,400
2,270
79.6
2014 annual release
724
590
Landfill
12,200
780
24.4
WIP
528
425
36.9
2,830
Waste water
2,320
1.72
STP
Over flow
108
74.5
584
1,190
70.3
1,860
4.72
Export
Recycling
Unconsidered
1,810
Use of BS
2,460
400
Air
142
43.0
0.35
180
N&U soil
977
6.10
ST soil
6,220
5.58
Surface Water
340
1,110
Sediment
12,200
Nano-Al2O3 (EU,2014)
d

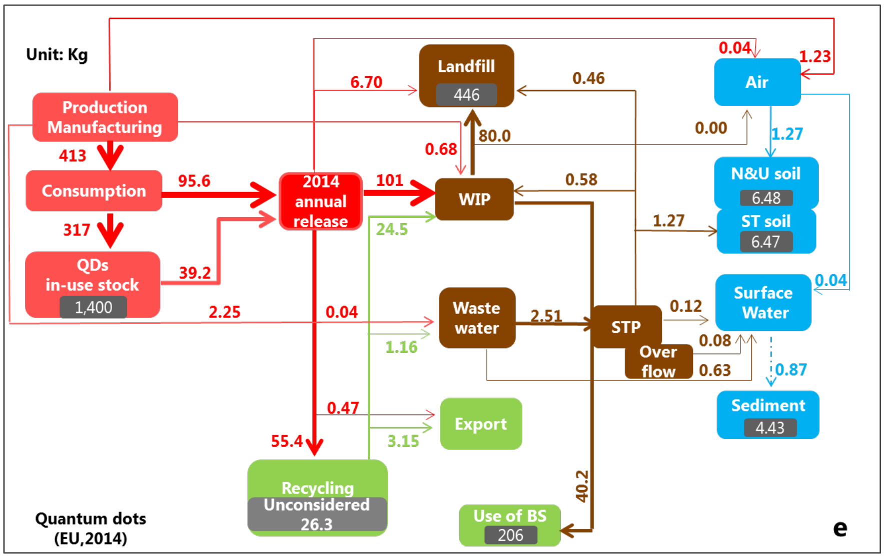


Figure A3.4. Mass flow diagrams for five ENMs in the EU in 2014 based on dynamic modeling: (a) nano-SiO_2, (b) nano-iron oxides, (c) nano-CeO_2, (d) nano-Al_2O_3, and (e) quantum dots. The red boxes represent the model input compartments. The green and brown boxes represent those waste treatment processes for which reuse or recycling processes are or are not involved, respectively. The blues boxes represent the environmental compartments. The dark grey boxes show the amount of ENMs held in stocks in each compartment up to 2014. A dashed line was used to represent the flow from surface water to sediment as two extreme scenarios were assumed: a) no transfer and b) complete sedimentation. All presented values are the means of the mass distributions and are rounded to three significant numbers. The unit for QDs is kg/y, while the unit for the other ENMs is tonnes/y. "Unconsidered" represents those products, of which the further behavior after recycling processes was not considered due to lack of data; WIP: waste incineration plant; STP: sewage treatment plant; BA: bottom ash; N&U soil: natural & urban soil. ST soil: sludge-treated soil.

The distribution of ENMs after release is determined by the processes that the nano-product is going through. For instance, QDs are mostly applied in a matrix within products and, consequently, only negligible amounts of QDs are released to wastewater during use. Another example is release of nano-CeO_2, of which about 97% is released to landfill and recycling at the EoL stage. This is resulting from the fact that around 88% of nano-CeO_2 are used in 'Polishing', 'Fuel additive, 'Glass', 'Paint', and 'Fuel cell' and only 0-5% from each category is released during use phase, mainly to wastewater (see Table S4b). The main use of nano-CeO_2 is in polishing slurries, and 90% of it is captured in the spent slurries, which are further treated and then completely landfilled (Ruth 2014). Another important use of nano-CeO_2 is in fuel additives, from which nano-CeO_2 is expected to be released with the exhaust after diesel combustion. However, the diesel particulate filter is able to capture 99% of particulate matters, which is finally collected for recycling (ATC 2013, Wade et al. 1981).

ENMs used in products that are likely to flow partially to wastewater (e.g. cosmetics, paints, and cleaning agents). As shown in Figure 4, there is a significant flow from consumption to wastewater for all ENMs except nano-CeO_2 and QDs. For other ENMs, between 20% and 33% are flowing to recycling. Waste incineration is another process to which large amounts of ENMs flow at the EoL stage. For nano-SiO_2, in particular, around 19% and 16% of ENMs flow to landfill and waste incineration plant, respectively. QDs have a notable percentage in medical use (in vitro biomarkers), consequently, three quarters of total released QDs is incinerated along with medical waste.

The outflows of ENMs from recycling was considered when relevant information was available (Caballero-Guzman et al. 2015). In Figure 4, the unconsidered stock inside the recycling box represents those product categories which are not covered by Caballero-Guzman et al (2016). Other studies that quantitatively cover the final disposal of ENM after recycling are not available. For all ENM except QDs, most of the mass flowing into recycling could not be further assessed. The work by Caballero-Guzman et al (2015) evaluated the fate of ENM in several product categories such as 'Paints', 'Textile' and 'Electronics'. These categories make up an important fraction for most of the ENM considered in this work (see Figure 2). However, ENM contained in paint have not yet reached the EoL stage in 2014 to a significant degree because of the long lifetime of buildings where paints are applied. Therefore, there is no substantial inflow from paint to recycling in 2014 for nano-SiO_2, nano-iron oxides, nano-CeO_2 and nano-Al_2O_3. The remaining flows into recycling are mainly from product categories not covered by the previous study and could therefore not be modelled (Caballero-Guzman et al. 2015). More than 70% of QDs are used in electronics and electrical devices and their flows after recycling were considered in the model. This is why the ratio between the outflow and inflow of the recycling of QDs is much higher than those of other ENMs.

A3.3.3. Predicted Environmental Concentrations

Table 2 gives the mean release concentrations of each ENM in technical and environmental systems in the seven geographical systems considered for the year 2014 (median, Q15, and Q85 values are presented in Table S6 of the SI). The concentrations in sink compartments (e.g. soils, sediments) reported in the previous static models referred to yearly increases in concentrations, while the concentrations in sinks reported in the current work are absolute ones considering the accumulation of ENMs over time (Gottschalk et al. 2009, Sun et al. 2014). Nano-SiO_2 with the largest use amount has the highest release concentrations in the environment, while QDs, with the smallest use amount, have the lowest concentrations. Nano-CeO_2, with smaller use amount than those of nano-iron oxides and nano-Al_2O_3, has nonetheless higher concentrations in landfilled waste due to the dominant release to landfill.

For all ENMs, the highest concentrations among the environmental compartments were found in sludge treated soil, which is at the level of mg/kg for nano-SiO_2, nano-iron oxides, and nano-Al_2O_3, µg/kg for nano-CeO_2, and ng/kg for QDs. The second highest ranked compartment is sediment followed by natural and urban soil, surface water, and, finally, air (ng/m^3 to fg/m^3). In the waste management processes, the highest concentrations of nano-CeO_2 is found/obtained in landfilled waste, while for all other ENMs the highest ones are predicted in sewage sludge. The lowest concentrations of all ENMs are found in sewage effluent. The ENMs in fly ash and bottom ash are a result of the incineration of waste and sludge. Hence, the higher concentrations in sewage sludge and in waste leads to a higher concentration in waste incineration bottom ash and fly ash.

Table A3.2. Concentrations of five ENMs in different environmental compartments and waste management processes in seven systems in 2014.

	Nano-SiO$_2$ concentration in 2014								Nano-CeO$_2$ concentrations in 2014							
Compartments	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**unit**	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**unit**
STP Effluent	65	74	51	48	23	34	12	**µg/L**	37	44	25	29	19	18	4.8	**ng/l**
STP Sludge	4.4	5.3	2.4	4.6	2.1	3.3	6.8	**g/kg**	1.8	2.5	0.89	2	1.2	1.3	2.6	**mg/kg**
Landfilled waste	490	620	450	1500	400	200	-	**mg/kg**	6.5	23	3.3	41	4.4	1.3	-	**mg/kg**
Incinerated waste	490	620	380	470	180	5900	670	**mg/kg**	8.4	10	8.1	7.8	4.7	150	10	**µg/kg**
Bottom ash	3.4	4.5	2.8	3.1	1.1	25	5.7	**g/kg**	0.55	0.85	0.66	0.25	0.32	7.1	0.9	**mg/kg**
Fly ash	4.7	6.1	3.9	4.1	1.5	35	7.9	**g/kg**	0.77	1.2	0.91	0.36	0.45	9.8	1.2	**mg/kg**
Air	16	34	8.7	2.6	10	18	48	**ng/m3**	42	100	20	6.8	35	43	110	**pg/m3**
N&U soil	86	170	52	28	68	92	290	**µg/kg**	42	96	22	14	39	41	120	**ng/kg**
ST soil	150	390	330	240	240	110	-	**mg/kg**	60	180	120	110	130	45	-	**µg/kg**
Surface water	4.3	4.4	2.5	0.22	8.6	11	4.2	**µg/L**	2.0	2.6	1.1	0.11	5.1	4.9	1.9	**ng/l**
Sediment	79	79	46	4.1	180	210	75	**mg/kg**	35	46	19	2.1	95	87	32	**µg/kg**

	Nano-iron oxides concentrations in 2014								Nano-Al$_2$O$_3$ concentrations in 2014							
Compartments	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**unit**	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**unit**
STP Effluent	1.5	1.3	1.2	0.85	0.7	0.56	0.24	**µg/L**	3.6	5.1	3.5	2.6	2.5	2.5	1.1	**µg/L**
STP Sludge	75	85	44	73	53	45	110	**mg/kg**	240	380	160	250	220	240	620	**mg/kg**
Landfilled waste	0.89	1.9	1.03	4.7	1.2	0.35	-	**mg/kg**	9.9	20	13	41	16	4.7	-	**mg/kg**
Incinerated waste	1	1.2	1.3	0.94	0.67	19	1.4	**mg/kg**	9.6	17	11	9.9	6.1	140	21	**mg/kg**
Bottom ash	22	31	32	11	14	240	42	**mg/kg**	93	190	120	64	55	870	300	**mg/kg**
Fly ash	32	44	44	16	19	340	59	**mg/kg**	130	260	170	88	77	1200	410	**mg/kg**
Air	0.23	0.53	0.15	0.04	0.25	0.24	0.76	**ng/m3**	1.2	3.7	0.88	0.21	1.6	1.9	6.5	**ng/m3**
N&U soil	0.32	0.67	0.22	0.11	0.33	0.31	1.2	**µg/kg**	1.3	4.1	1.2	0.51	1.9	2	8.4	**ug/kg**
ST soil	2.7	6.6	6	4.2	5.8	1.7	-	**mg/kg**	8.2	27	24	14	23	7.9	-	**mg/kg**
Surface water	86	110	58	4.7	250	180	110	**ng/L**	0.31	0.53	0.22	0.02	1.1	1	0.65	**µg/L**
Sediment	1.6	2	1.03	0.09	4.4	3.6	2	**mg/kg**	5.7	9.3	4.1	0.33	19	17	11	**mg/kg**

	QD concentrations in 2014							
Compartments	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**unit**
STP Effluent	2.7	3.2	2.4	2.3	1.6	1.6	0.48	**pg/L**
STP Sludge	0.18	0.23	110	0.22	0.14	0.15	0.26	**µg/kg**
Landfilled waste	91	330	57	660	70	22	-	**ng/kg**
Incinerated waste	1.6	1.2	3.4	1.2	1.8	72	0.83	**µg/kg**
Bottom ash	8	6.1	17	6.1	9.3	260	4.8	**µg/kg**
Fly ash	11	8.4	23	8.5	13	350	6.6	**µg/kg**
Air	7.9	19	5.1	1.5	8.4	10	22	**fg/m3**
N&U soil	8.4	18	5.8	3	9.1	9.9	27	**pg/kg**
ST soil	6.2	17	16	11	15	5.3	-	**ng/kg**
Surface water	170	180	120	9.6	500	530	150	**fg/L**
Sediment	3.2	3.2	2.1	0.17	9.1	9.4	2.8	**ng/kg**

N&U soil is natural and urban soil, ST soil is sludge treated soil. EU: EU 27, CH: Switzerland, CE: Central Europe, EE: Eastern Europe, SE: Southern Europe, NE: Northern Europe, SEE: South-eastern Europe.

A3.3.4 Geographical distribution

Table 2 shows that regional differences of the release concentrations in air, natural and urban soil, surface water and sediments of each ENM follow the same patterns, which are driven by the mass or volume of the receiving medium of each compartment. However, concentrations in sludge treated soil are dependent on the concentrations in sewage sludge, the amount of sewage sludge that is applied as fertilizer and the volume of sludge treated soil. Nano-SiO_2 is presented as an example in Figure 5, which shows the regionalized distributions of nano-SiO_2 in 2014 in natural and urban soil, sludge treated soil, surface water, and sediment. Each subplot comprises one large map and one small map, where the large maps shows the distribution of nano-SiO_2 in four compartments in the different regions and the small maps show the average concentration levels in the EU. As shown in Figure 5a, Switzerland has the highest concentration of nano-SiO_2 in natural and urban soil (290 µg/kg), due to its relatively large use amount and small geographical area, while Northern Europe has the lowest concentration (28 µg/kg), due to the relatively small use amount within the region and its large geographical area. The second highest concentration in natural and urban soil was found in Central Europe, followed by South-eastern Europe, EU, and Southern European Eastern Europe.

The lowest concentration in sludge treated soil is observed in South-eastern Europe (from 5.3 ng/kg for QDs to 110 mg/kg nano-SiO_2), as it is the region with the lowest percentage of sludge applied to soil. Central Europe has the highest concentration of ENMs in sludge treated soil, a consequence of having the highest concentration of ENMs in sewage sludge. However, the EU, despite the largest amount of sludge applied in agriculture (see Table S5 and S7), has only the second lowest concentration due to the low concentrations of ENMs in the sludge, as well as the large area of sludge treated soil in the region. There is no land application of sludge in Switzerland. Hence, Switzerland is blank in the map in Figure 5b.

The most relevant flow to surface water comes from untreated wastewater, as shown in Figure 4. Therefore, the fraction of households in a region that is not connected to the sewage treatment system is the decisive factor in determining the amount of ENM that are released to surface water. This is especially noticeable for nano-SiO_2 because a large share of it is used in cleaning agents and cosmetics, with the resulting release of nano-SiO_2 to waste water. As a result of the lowest share of households connected to sewage treatment plants, South-eastern Europe has the highest concentration of nano-SiO_2 in surface water (11 µg/L). This is approximately 50 times greater than the concentration in Northern Europe. Although the connection to wastewater systems in Northern Europe is not as high as it is in Central Europe, it has the lowest concentration in surface water (0.22 µg/L) because it has the largest volume of surface water amongst all EU sub-systems. The ranking of regions with regards to the concentrations of ENM in sediment is the same as that for concentrations in surface water. This is caused by the scenario-type modeling with complete sedimentation for calculating the concentration for the sediment compartment, and no sedimentation when calculating the surface water concentrations.

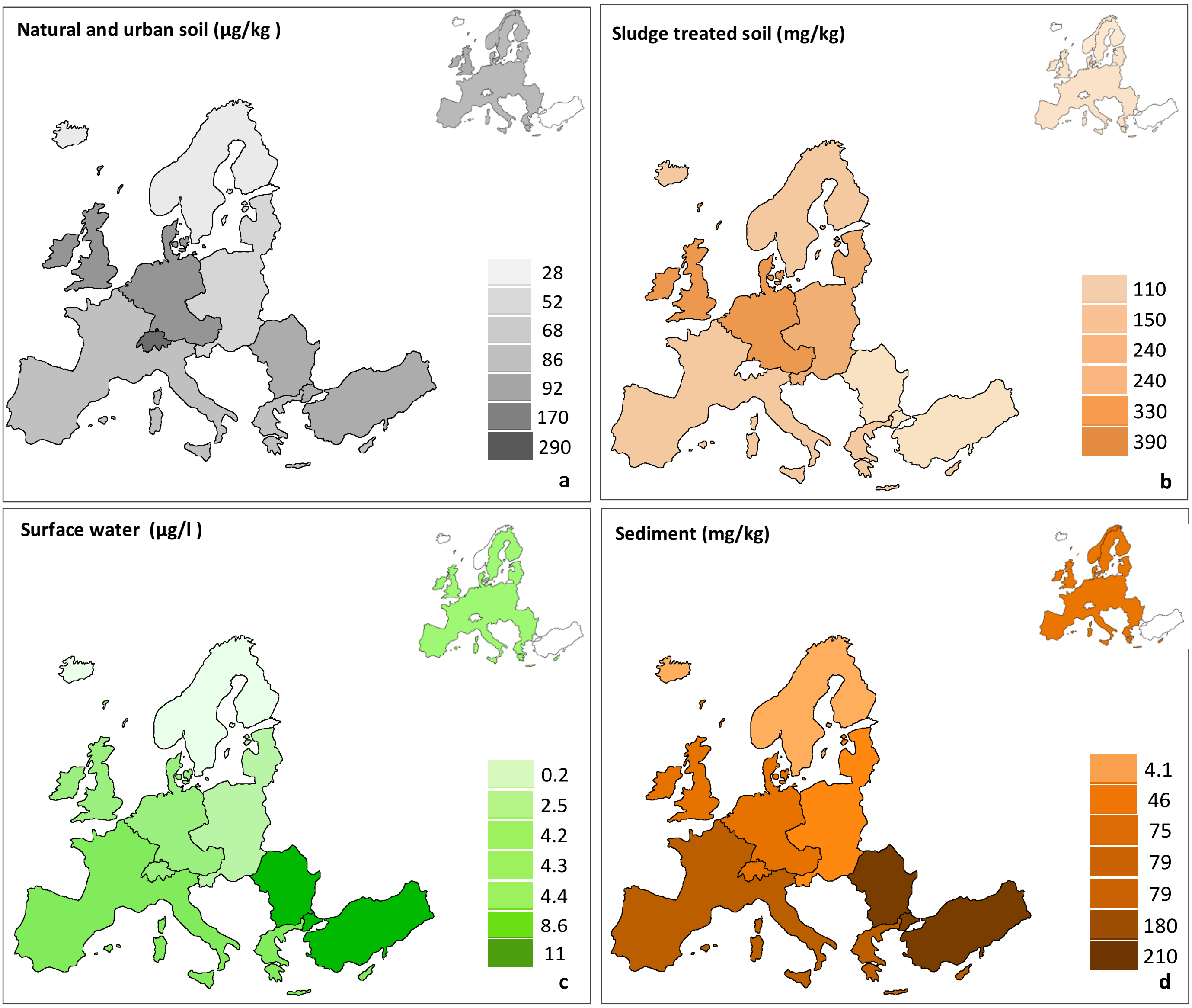


Figure A3.5. Geographical distribution of nano-SiO_2 in seven European regions. The darkness of a color is defined by the concentration in each compartment among all regions. The darkest and lightest colors indicate the highest and lowest concentrations, respectively. Each plot comprises one large map and one small map, presented in the upper right corner. The small maps indicate the average concentration in the EU.

4.4 Discussion

Many research studies have already endeavoured to estimate the release of ENMs towards the environment (Boxall et al. 2007, Gottschalk et al. 2015, Johnson et al. 2011, Keller &Lazareva 2014, Sun et al. 2016, Wang et al. 2016a). The predicted concentrations for nano-SiO_2, nano-CeO_2, and nano-Al_2O_3 in the current study are one to three orders of magnitudes higher than those in Boxall et al (2007), which is due to the fact that they considered only single nano-products for each ENM. For nano-CeO_2 and quantum dots, the predicted release concentrations in Gottschalk et al (2015) are ten to hundred times lower compared with our study. The reason of the differences is that no direct environmental release of these two ENMs was estimated in the previous model (Gottschalk et al. 2015). Compared to the output of our model, the concentrations in effluents and biosolids from sewage treatment plants around San Francisco Bay predicted by Keller et al. (2014) are one to three orders of magnitude higher for nano-iron oxides and CeO_2, one order of magnitude lower for nano-SiO_2, and at the same level for nano-Al_2O_3. These differences can be explained by the considered product categories, and the release pathway of which varies from the current work.

The mass flow results from our static modelling work of nano-iron oxides have the same order of magnitude with those in the current work (Wang et al. 2016a). The estimated use volume in the current study is more than six times higher than in the static study because additional data sources were considered, however the released amount in the EU is close to our previous study (Wang et al. 2016b). This similarity despite very different input is the effect of considering time-dependent input, use release and EoL release in the current study. Specifically, EoL release did not occur for the product category 'Paint' in 2014 because it is still stored in the in-use stock, whereas the EoL release of ENMs in the static model occurs in the production year of the ENMs and the nano-products. Therefore, the predicted concentrations in this work are more complete by considering a more complex system with time-dependent variables. The gap between the annual input and the annual release and the high percentage of release from the in-use stock for some materials highlights the importance of using dynamic compared to static MFA, which does not consider the in-use stock.

The large data set of five ENMs in various compartments in seven regions obtained in our work can be evaluated for the relationship between use amount and the resulting release concentrations in environmental compartments. Figure 6 shows that the concentration levels of ENMs in environmental compartments are, as expected, generally correlated to the input of the ENMs within the system boundary. However, there is considerable deviation from a linear relationship because of the different life cycles of the nano-products from which varying proportions of ENMs are released into different compartments. This is especially visible within the landfill process: higher concentrations of nano-CeO_2 are predicted in landfilled waste compared to nano-iron oxides, even though the use level of nano-CeO_2 is lower. This is mainly due to its important use in mechanical chemical polishing, which results in a dominant release to landfill. Previous studies assumed a different release pathway for nano-CeO_2, namely that nano-CeO_2 is released completely from the polishing slurry to air and water (Collin et al. 2014). However, the spent polishing slurry is, in fact, collected and then landfilled (Janos et al. 2015, Ruth 2014). The current recycling technology is not yet advanced enough to allow an industrial scale recycling of nano-CeO_2 from spent polishing slurry. In our model, around 75% of nano-CeO_2 is predicted to be eventually released to landfills. This result is close to that of a previous modeling study (Keller &Lazareva 2014) in which 80-97% of nano-CeO_2 was suggested to flow to landfills. However, this study considered a very general "electronics and optics” category without specifying any particular use. The second main use of nano-CeO_2 is in diesel additives. Park et al (2008) have estimated a release of 6-100% of nano-CeO_2 to air under different scenarios, which are: 1) no filter, 2) traps, 3) filter equipped in vehicles. However, due to the technological advancement (ATC 2013), as well as the implementation of legislation (PARLIAMENT 2007), modern diesel vehicles are equipped with particle filters, which are able to remove 99% of particulate matter from the exhaust. The captured particles are finally entering recycling at the EoL of the filter (ATC 2013). Therefore, in the current study an estimation of 99% of nano-CeO_2 flowing from fuel additive to recycling is much closer to the real situation.

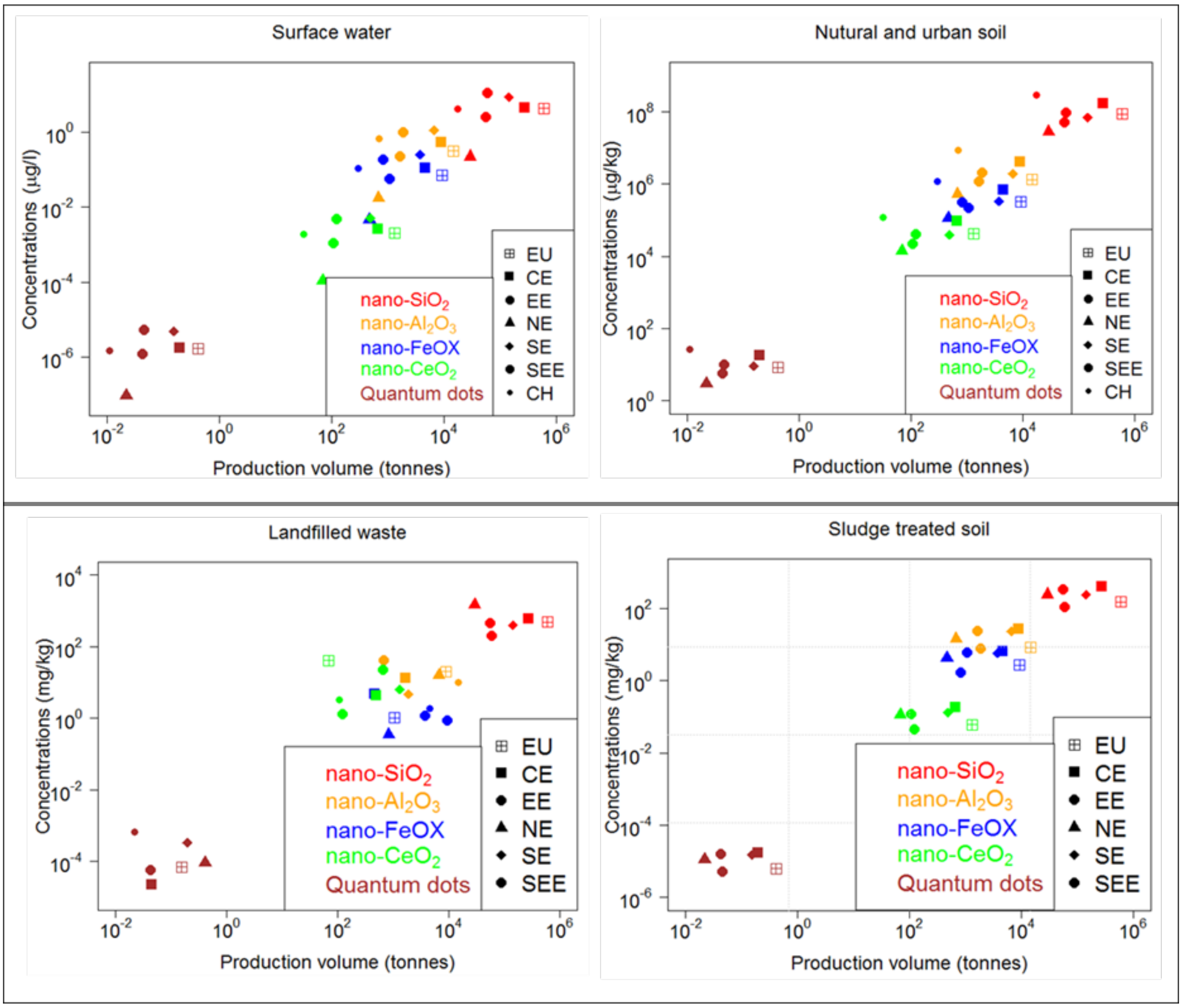


Figure 4-6. Correlation between production volume of ENMs and their concentrations in different regions in (a) surface water; (b) natural and urban soil; (c) landfill waste; (d) sludge treated soil. The colours represent the ENM, the shape represent the region.

The release distributions of other ENMs during the use phases and EoL stages are different from nano-CeO_2. For nano-iron oxides and nano-Al_2O_3, the two main release flows are to wastewater due to use, mainly in the product categories ‘Cosmetics’, ‘Medical use’ or ‘Cleaning agent’, and to recycling, mostly resulting from applications in the product categories ‘Catalyst’, ’Electronics’, or ‘Special paper’. ENMs are largely removed during sewage treatment and are then partially transferred to soil by sludge application. Therefore, the main flows of nano-iron oxides and nano-Al_2O_3 go to recycling and soil. Nano-SiO_2 has two other release destinations in addition to recycling and wastewater: landfills and bottom ash from incineration plants. This is due to a significant amount used in ‘Plastic’ with the major part of the EoL plastic products ending up in landfills and incineration plants with the municipal waste (Eurostat 2017). Keller et al (2014) have considered similar product categories for these three ENMs: ‘Catalyst’, ‘Personal care’, ‘Medical use’, ‘Paint’, and ‘Electronics’, but they modelled release mainly to landfill and soil. Moreover, their study did not consider any recycling processes. Two other static MFA modelling studies for nano-SiO_2 and nano-iron oxides have used similar release pathways as in the current work (Wang et al. 2016a, Wang et al. 2016b). However, the flows of nano-SiO_2 and nano-iron oxides to landfill in these studies accounted for a higher percentage of the total release, which can be explained by the high percentage of ENMs used in paint of which 40% are released to landfill in the previous model. Additional data taken into consideration in the current study affected the distribution of the produced mass to product categories. This shows the great importance to obtain specific data about product distribution, a parameter that exhibits a large variability between different sources. The approach chosen in this work was to combine different sources and derive an overall probability distribution. Quantum dots have about 25% use in medical applications which results in release to incineration plants together with medical waste (ICRC 2011). The remainder is mostly used within electronic devices, from which only release during recycling is likely. This is consistent with the study from Gottschalk et al (2015), where they estimated that 90% of quantum dots are ending up in recycling and 10% go to incineration plants.

Previous studies have shown that the local waste management systems have a strong influence on the distribution of ENMs (Heggelund et al. 2016, Keller &Lazareva 2014, Wigger et al. 2015). Also in the current work, we found that the regional waste management processes determine the distribution of EoL products between the waste management processes, and ultimately, determine the flows of ENMs into different environmental compartments. For instance, the proportion of wastewater treated in each region affects the release of ENMs to surface water and the concentration of ENMs in sewage sludge, which, together with the amount of sludge applied to land (depending on the national regulation) and the area size of sludge treated soil, determines the concentration of ENMs in sludge treated soil. Another factor affecting the concentration level is the volume or mass of the receiving medium. Therefore, Northern Europe was found to have the lowest concentration of ENMs in most compartments because it has the lowest ratio between input of ENMs and the volume/mass of the receiving medium in each compartment. Southeastern Europe had the highest concentration of ENMs in surface waters because of a high share of wastewater bypassing wastewater treatment. The previous regionalized modelling study for Europe from Dumont et al (2015) considered regional differences of the water compartment but did not include differences in the regional waste management. Their study also predicted the lowest and highest concentrations in Northern Europe and Southern Europe, respectively. The differences in regional concentrations demonstrate that subdividing Europe according to the similarity of waste management systems of countries enabled us to better estimate the distribution of ENMs between waste management processes within Europe. However, other studies have shown that concentration of ENMs can vary a lot at even higher spatial resolution within a single watershed (Dale et al. 2015b, Dumont et al. 2015, Gottschalk et al. 2011). The predicted concentrations in each geographical system in this work represent the average values of the region. However, concentrations at local level might be under- or over-estimated because concentrations at local release points, such as effluent of STP, are more likely higher than those in rural areas. The results still provide useful information for further investigating the local concentrations in the environmental compartments.

After being released to the environment, ENMs are unavoidably affected by physical and chemical reactions, such as homo and heteroagglomeration, sedimentation and transformations (Arvidsson et al. 2011). These processes are not included in our model and therefore it is not able to provide real environmental concentrations. The model is quantifying the total amount of ENM that end up in an environmental compartment but this may not represent the actually present amount, especially for reactive ENM such as QD. We therefore use the term “release concentration” to distinguish our values from environmental concentrations derived from fate models. By considering fate processes, environmental fate models are able to predict the distribution and concentration of different species of ENMs (Liu &Cohen 2014). These fate modelling studies are based on the input data from release models (Markus et al. 2017). For example, the amount of nano-Ag and nano-ZnO loading to the environmental systems in the fate model of Dale et al (2015) are taken from the release studies of Gottschalk et al (2009) and Keller et al (2014). The output of Keller et al (2014) has been also used by Pu et al (2016) for the fate modeling of nano-Cu in 17 subcontinental freshwaters.

A validation of the modelling output (flows and concentrations) by measurements is not possible at the moment. It has been discussed in detail in Nowack et al. (2015) that the results currently provided by analytical methods are not yet specific to ENM (they include all nanoparticles) or are just not sensitive enough to reach realistic environmental concentrations. The available analytical results can therefore not be used to validate model results but they provide additional information that can be used together with the modelling results to gain a complete understanding of the presence of natural and engineered nanoparticles in the environment.

The uncertainty of the current modelling results is due to incomplete knowledge regarding input parameters e.g. production/use volume, product allocation, release schedules of ENMs from products and the distribution to the receiving compartments after release. The probabilistic approach used in this study allowed us to include the plausible values for each parameter and represent uncertain input parameters. The dynamic aspect considered the impact of time-dependent variables on outcome of the model by incorporating use amounts over multiple years into the model input and the dynamic release of ENMs from nano-products. In the current model, we assumed that ENM application share and release schedule would remain constant over the 31 year time period. This is a limitation of the model as it is unlikely that ENM application shares and release schedules remained constant in the past. Sun et al (2017) have predicted future flows of ENMs toward the environment in the EU under various application scenarios of ENMs in products. They identified a notable difference in mass flows over time due to the change of the share of ENMs in product categories. DPMFA models can be improved by considering the development in ENMs application and release schedules.

A further limitation of this work is that the destinations to which nano-products flow following recycling were not considered fully for all product categories. However, even the partial consideration of flows out of recycling constitutes a significant advancement over previous modelling studies that either did not consider recycling at all (Gottschalk et al. 2015, Keller &Lazareva 2014, Mueller &Nowack 2008) or just included flows into recycling but no outflows (Sun et al. 2014). For all considered ENMs, 20%-58% of the total release in 2014 flows to recycling. These EoL products are further shredded (electronics), cut (composites) or dissolved (paper), during which a release to air or water might occur (Methner et al. 2012, Raynor et al. 2012, Zhang et al. 2016). The remaining materials are further landfilled, incinerated, or enter recycling processes. However, there is no published data available for every product category to further investigate where the ENMs flow after EoL treatment. In this study, the behaviour of ENMs during recycling was included for only two categories, ‘Paint’ and ‘Electrical’ based on the data provided by Caballero-Guzman et al (2015). Given the significant amount of ENMs flowing to recycling, a better understanding of the quantities and destinations of ENMs outputs from recycling is really important to improve estimates of ENM flows during the EoL stage.

4.5 Conclusions

Our work provides the most up-to-date information on the flows of nano-SiO_2, nano-iron oxides, nano-CeO_2, nano-Al_2O_3, and QDs to the environment. The dynamic nature of the model allows to consider the accumulation of ENM in stocks and environmental sinks and is therefore more accurate that results from static models. The modeling is based on an integration of all available data on production of ENM, use of nanoproducts and flows to technical systems. The probabilistic nature of the model allows to capture quantitatively the full range of uncertainty. Large shares of the total flows go to waste treatment systems and thus this part of the model was considered in more detail. Because waste handling exhibits large differences between European countries, a regionalization was included in the model.

The flows derived from our material flow model can be used as input values in future exposure and fate modeling of ENMs at the sub-regional/local level. Environmental risk assessment studies for ENMs also rely on exposure data provided by mass flow modelling studies (Coll et al. 2016, Gottschalk &Nowack 2012). The current study can therefore also lay the foundation for a comprehensive assessment of the environmental risk posed by the investigated ENMs, which subsequently may support policy-making.

A3.5 References

Arvidsson R, Molander S, Sanden BA, Hassellov M (2011): Challenges in Exposure Modeling of Nanoparticles in Aquatic Environments. Human and Ecological Risk Assessment 17, 245-262

ATC 2013: Fuel Additives: Use and Benefits

Baalousha M, Cornelis G, Kuhlbusch TAJ, Lynch I, Nickel C, Peijnenburg W, van den Brink NW (2016): Modeling nanomaterial fate and uptake in the environment: current knowledge and future trends. Environmental Science-Nano 3, 323-345

Bornhoft NA, Sun TY, Hilty LM, Nowack B (2016): A dynamic probabilistic material flow modeling method. Environmental Modelling & Software 76, 69-80

Boxall ABA, Chaudhry Q, Sinclair C, Jones AD, Aitken R, Jefferson B, Watts C 2007: Current and future predicted environmental exposure to engineered nanoparticles, Central Science Laboratory, Sand Hutton, UK

Caballero-Guzman A, Sun T, Nowack B (2015): Flows of engineered nanomaterials through the recycling process in Switzerland. Waste Manag 36, 33-43

Caballero-Guzman A, Nowack B (2016): A critical review of engineered nanomaterial release data: Are current data useful for material flow modeling? Environ Pollut 213, 502-17

Chen Z, Yadghar AM, Zhao L, Mi ZT (2011): A review of environmental effects and management of nanomaterials. Toxicological and Environmental Chemistry 93, 1227-1250

Coll C, Notter D, Gottschalk F, Sun T, Som C, Nowack B (2016): Probabilistic environmental risk assessment of five nanomaterials (nano-TiO_2, nano-Ag, nano-ZnO, CNT, and fullerenes). Nanotoxicology 10, 436-44

Collin B, Auffan M, Johnson AC, Kaur I, Keller AA, Lazareva A, Lead JR, Ma XM, Merrifield RC, Svendsen C, White JC, Unrine JM (2014): Environmental release, fate and ecotoxicological effects of manufactured ceria nanomaterials. Environmental Science-Nano 1, 533-548

Dale AL, Casman EA, Lowry GV, Lead JR, Viparelli E, Baalousha M (2015a): Modeling Nanomaterial Environmental Fate in Aquatic Systems. Environmental Science & Technology 49, 2587-2593

Dale AL, Lowry GV, Casman EA (2015b): Stream Dynamics and Chemical Transformations Control the Environmental Fate of Silver and Zinc Oxide Nanoparticles in a Watershed-Scale Model. Environmental Science & Technology 49, 7285-7293

Dumont E, Johnson AC, Keller VDJ, Williams RJ (2015): Nano silver and nano zinc-oxide in surface waters - Exposure estimation for Europe at high spatial and temporal resolution. Environ. Pollut. 196, 341-349

ECHA 2016: Guidance on Information Requirements and Chemical Safety Assessment Chapter R.16: Environmental Exposure Assessment

EEA. EEA 2013: Urban Waste Water Treatment, Kongens Nytorv 6, 1050 Copenhagen K Denmark

European Commission 2010: Europe in Figures. Eurostat Yearbook 2010. Luxembourg: Publications Office of the European Union

Eurostat (2014): Treatment of Waste by Waste Category, Hazardousness and Waste Operations-- Mineral Waste from Construction and Demolition

Eurostat (2017): Municipal Waste Stastistic- mixed waste

Future Markets (2014): The Global Market for Metal Oxide Nanoparticles 2014. Future Market

Gottschalk F, Sonderer T, Scholz RW, Nowack B (2009): Modeled environmental concentrations of engineered nanomaterials (TiO_2, ZnO, Ag, CNT, fullerenes) for different regions. Environ. Sci. Technol. 43, 9216-9222

Gottschalk F, Scholz RW, Nowack B (2010): Probabilistic material flow modeling for assessing the environmental exposure to compounds: Methodology and an application to engineered nano-TiO2 particles. Environ. Modeling Software 25, 320-332

Gottschalk F, Ort C, Scholz RW, Nowack B (2011): Engineered nanomaterials in rivers - exposure scenarios for Switzerland at high spatial and temporal resolution Environ. Pollut. 159, 3439-3445

Gottschalk F, Nowack B (2012): A probabilistic method for species sensitivity distributions taking into account the inherent uncertainty and variability of effects to estimate environmental risk. Integrated Environmental Assessment and Management 9, 79–86

Gottschalk F, Lassen C, Kjoelholt J, Christensen F, Nowack B (2015): Modeling flows and concentrations of nine engineered nanomaterials in the Danish environment. Int J Environ Res Public Health 12, 5581-602

Heggelund L, Hansen SF, Astrup TF, Boldrin A (2016): Semi-quantitative analysis of solid waste flows from nano-enabled consumer products in Europe, Denmark and the United Kingdom - Abundance, distribution and management. Waste Management 56, 584-592

Hendren CO, Lowry M, Grieger KD, Money ES, Johnston JM, Wiesner MR, Beaulieu SM (2013): Modeling Approaches for Characterizing and Evaluating Environmental Exposure to Engineered Nanomaterials in Support of Risk-Based Decision Making. Environmental Science & Technology 47, 1190-1205

ICRC 2011: Medical Waste Management

Janos P, Kuran P, Ederer J, Stastny M, Vrtoch L, Psenicka M, Henych J, Mazanec K, Skoumal M (2015): Recovery of Cerium Dioxide from Spent Glass-Polishing Slurry and Its Utilization as a Reactive Sorbent for Fast Degradation of Toxic Organophosphates. Advances in Materials Science and Engineering 2015, 8

JNIES (2016): Lifespan database for Vehicles, Equipment, and Structures: LiVES. Japanese National Institute for Environmetal Studies (JNIES)

Johnson AC, Bowes MJ, Crossley A, Jarvie HP, Jurkschat K, Jorgens MD, Lawlor AJ, Park B, Rowland P, Spurgeon D, Svendsen C, Thompson IP, Barnes RJ, Williams RJ, Xu N (2011): An assessment of the fate, behaviour and environmental risk associated with sunscreen TiO2 nanoparticles in UK field scenarios. Science of the Total Environment 409, 2503-2510

Keller AA, McFerran S, Lazareva A, Suh S (2013): Global life cycle releases of engineered nanomaterials. Journal of Nanoparticle Research 15, 17

Keller AA, Lazareva A (2014): Predicted Releases of Engineered Nanomaterials: From Global to Regional to Local. Environmental Science & Technology Letters 1, 65-70

Keller AA, Vosti W, Wang HT, Lazareva A (2014): Release of engineered nanomaterials from personal care products throughout their life cycle. Journal of Nanoparticle Research 16

Klaine SJ, Koelmans AA, Horne N, Carley S, Handy RD, Kapustka L, Nowack B, von der Kammer F (2012): Paradigms to assess the environmental impact of manufactured nanomaterials. Environ. Toxicol. Chem. 31, 3-14

Koivisto AJ, Jensen ACØ, Kling KI, Nørgaard A, Brinch A, Christensen F, Jensen K, A (2017): Quantitative material releases from products and articles containing manufactured nanomaterials: Towards a release library. NanoImpact 5, 139-142

Liu HH, Cohen Y (2014): Multimedia Environmental Distribution of Engineered Nanomaterials. Environmental Science & Technology 48, 3281-3292

Mahapatra I, Sun TY, Clark JRA, Dobson PJ, Hungerbuehler K, Owen R, Nowack B, Lead J (2015): Probabilistic modelling of prospective environmental concentrations of gold nanoparticles from medical applications as a basis for risk assessment. Journal of Nanobiotechnology 13, 1-14

Markus AA, Parsons JR, Roex EWM, de Voogt P, Laane R (2017): Modelling the Release, Transport and Fate of Engineered Nanoparticles in the Aquatic Environment - A Review. Rev Environ Contam Toxicol 243, 53-87

Methner M, Crawford C, Geraci C (2012): Evaluation of the Potential Airborne Release of Carbon Nanofibers During the Preparation, Grinding, and Cutting of Epoxy-Based Nanocomposite Material. Journal of Occupational and Environmental Hygiene 9, 308-318

Money ES, Reckhow KH, Wiesner MR (2012): The use of Bayesian networks for nanoparticle risk forecasting: Model formulation and baseline evaluation. Science of the Total Environment 426, 436-445

Mueller NC, Nowack B (2008): Exposure modeling of engineered nanoparticles in the environment. Environ Sci Technol 42, 4447-53

Nowack B, Baalousha M, Bornhoft N, Chaudhry Q, Cornelis G, Cotterill J, Gondikas A, Hassellov M, Lead J, Mitrano DM, von der Kammer F, Wontner-Smith T (2015): Progress towards the validation of modeled environmental concentrations of engineered nanomaterials by analytical measurements. Environmental Science-Nano 2, 421-428

Nowack B (2017): Evaluation of environmental exposure models for engineered nanomaterials in a regulatory context NanoImpact

O'Brien N, Cummins E (2008): Recent developments in nanotechnology and risk assessment strategies for addressing public and environmental health concerns. Human and Ecological Risk Assessment 14, 568-592

PARLIAMENT E 2007: on type approval of motor vehicles with respect to emissions from light passenger and commercial vehicles (Euro 5 and Euro 6) and on access to vehicle repair and maintenance information

Pu YB, Tang F, Adam PM, Laratte B, Ionescu RE (2016): Fate and Characterization Factors of Nanoparticles in Seventeen Subcontinental Freshwaters: A Case Study on Copper Nanoparticles. Environmental Science & Technology 50, 9370-9379

Raynor PC, Cebula JI, Spangenberger JS, Olson BA, Dasch JM, D'Arcy JB (2012): Assessing Potential Nanoparticle Release During Nanocomposite Shredding Using Direct-Reading Instruments. Journal of Occupational and Environmental Hygiene 9, 1-13

Ruth J 2014: Post use options for CMP waste, Grenoble

SRIConsulting 2011: The Chemical Economics Handbook: Inorganic chemical pigments

Sun TY, Gottschalk F, Hungerbühler K, Nowack B (2014): Comprehensive Modeling of Environmental Emissions of Engineered Nanomaterials. Environ. Pollut. 185, 69-76

Sun TY, Conroy G, Donner E, Hungerbuhler K, Lombi E, Nowack B (2015): Probabilistic modelling of engineered nanomaterial emissions to the environment: a spatio-temporal approach. Environmental Science-Nano 2, 340-351

Sun TY, Bornhoft NA, Hungerbuhler K, Nowack B (2016): Dynamic Probabilistic Modeling of Environmental Emissions of Engineered Nanomaterials. Environmental Science & Technology 50, 4701-4711

Sun TY, Mitrano DM, Bornhöft NA, Scheringer M, Hungerbühler K, Nowack B (2017): Envisioning Nano Release Dynamics in a Changing World: Using Dynamic Probabilistic Modeling to Assess Future Environmental Emissions of Engineered Nanomaterials. Environmental Science & Technology 51, 10

Wade WR, White JE, Florek JJ (1981): Diesel Particulate Trap Regeneration Techniques. SAE International, pp. Paper No. 810118

Wang Y, Deng L, Caballero-Guzman A, Nowack B (2016a): Are engineered nano iron oxide particles safe? an environmental risk assessment by probabilistic exposure, effects and risk modeling. Nanotoxicology 10, 1545-1554

Wang Y, Kalinina A, Sun T, Nowack B (2016b): Probabilistic modeling of the flows and environmental risks of nano-silica. Science of the Total Environment 545-546, 67-76

Wigger H, Hackmann S, Zimmermann T, Koser J, Thoming J, von Gleich A (2015): Influences of use activities and waste management on environmental releases of engineered nanomaterials. Science of the Total Environment 535, 160-171

Wohlleben W, Neubauer N (2016): Quantitative rates of release from weathered nanocomposites are determined across 5 orders of magnitude by the matrix, modulated by the embedded nanomaterial. NanoImpact 1, 39-44

Zhang H, Bussini D, Hortal M, Elegir G, Mendes J, Beneyto MJ (2016): PLA coated paper containing active inorganic nanoparticles: Material characterization and fate of nanoparticles in the paper recycling process. Waste Management 52, 339-345

Annex 4. Risk assessment of engineered nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3 and quantum dots

Submitted to Environmental Toxicology and Chemistry

Abstract

A lot of research studies have endeavored to investigate the ecotoxicological hazards of engineered nanomaterials (ENM). However, little is known regarding the actual environmental risks of ENM, combining both hazard and exposure data. The aim of this study is to quantify the environmental risks for nano-Al_2O_3, nano-SiO_2, nano iron oxides, nano-CeO_2, and quantum dots by comparing the predicted environmental concentrations (PEC) with the predicted no effected concentrations (PNEC). The PEC values of these five ENMs in fresh waters in 2020 for northern Europe and southeastern Europe were taken from a published dynamic probabilistic material flow analysis model. PNEC values were calculated using probabilistic species sensitivity distribution (PSSD). The order of the PNEC values was quantum dots<nano-CeO_2<nano iron oxides<nano-Al_2O_3<nano-SiO_2. The risks posed by these five ENMs were demonstrated to be in the reverse order: nano-SiO_2>nano-Al_2O_3>nano iron oxides>nano-CeO_2>quantum dots. However, all risk characterization values are three to seven orders of magnitude lower than one and no risk was therefore predicted for any of the investigated ENMs at the estimated release level in 2020. Compared to static models, the dynamic material flow model allowed us to use PEC values based on a more complex parameterization, considering a dynamic input over time and time-dependent release of ENMs. The PSSD approach makes it possible to include all available data to estimate hazards of ENMs by considering the whole range of variability between studies and material types. The risk assessment approach is therefore able to handle the uncertainty and variability associated with the collected data. The results of the current study are able to provide a scientific foundation for risk-based regulatory decisions of the investigated ENM.

A4.1 Introduction

Applications of engineered nanomaterials (ENMs) are ubiquitous in industry and consumer products due to the rapid development of nanotechnology [220, 221]. The use of these nanoproducts inevitably causes the release of ENMs throughout the whole life cycle of nanoproducts to air, soil, water and sediments [222]. It has been demonstrated that ENMs often have special properties, which are more likely to induce hazardous effects compared to conventional materials [92, 223]. Therefore, it is imperative to evaluate the environmental risks posed by these materials, especially to support regulatory decision making [224]. The quantitative environmental risk assessment approach is comparing the exposure level of a compound with their hazard by dividing the predicted environmental concentration (PEC) by the predicted no effect concentration (PNEC) [225].

With the current analytical techniques it is difficult to identify specifically engineered nanoparticles in the environment [30]. This therefore limits the exposure assessment and consequently restricts the environmental risk evaluation of ENMs [33]. Currently, environmental exposure assessments are mostly depending on an exposure modelling approach [18]. Environmental exposure models have been advancing significantly during the last years, going from simplified to more complex models, from deterministic to probabilistic modeling, from regional to local systems, and from static to dynamic models [18, 187, 226]. Gottschalk et al (2010) developed a probabilistic material flow analysis (PMFA) model based on a life-cycle perspective, which is able to include all applications of a specific ENM and handle the uncertainty and variability associated with the input parameters, e.g. production volume or product allocation of ENMs. Based on the same concept, a dynamic probabilistic material flow analysis (DPMFA) model was developed by Bornhöft et al (2016) and used to estimate the environmental release and concentrations of nano-TiO_2, nano-Ag, nano-ZnO, and CNT in the EU in different scenarios [46, 47]. Another study has predicted environmental concentrations of nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots in seven European regions using the same DPMFA model [227]. Compared to the static model, the DPMFA model considered 1) a dynamic input of ENMs to the system from 1990 to 2020, 2) a dynamic release: ENM release was considered to follow different kinetics for different product categories, resulting in different delayed release schedules, and 3) the differences in waste management systems among European regions. These novel aspects make the PEC values from the dynamic models more realistic than those from the simpler static models.

The other critical point of environmental risk assessment is the hazard evaluation. The species sensitivity distribution method (SSD) is able to derive a PNEC, defined as the 5th percentile of the SSD, a threshold concentration so that 95% of species in an ecosystem are protected [225]. SSD is a well-established way to estimate the adverse effects of chemical substances by considering all available data instead of just using the lowest response endpoint [51-53]. The challenges of assessing the ecotoxicity of ENMs using the SSD method are manifold: There is only limited availability of ecotoxicological studies: whereas the effects on aquatic organisms have been intensively studied, much less studies are available for soils and sediments. The observed toxic effects also show a very large variability between studies, due to differences in test conditions and the different forms of the ENM that are investigated [54]. To overcome those challenges and handle variability of data, Gottschalk et al (2012) developed a probabilistic species sensitivity distribution (PSSD) method, which was applied later to evaluate hazardous effect of five ENMs (nano-TiO_2, nano-Ag, nano-ZnO, fullerenes and CNT) [55]. Semenzin et al. (2015) assessed the ecotoxicity of nano-TiO_2 using a weighted SSD approach, which includes three weighting criteria: the species relevance, the trophic level abundance and the nanotoxicity data quality. Garner et al. (2015) combined all available data to investigate toxicity of nano-Ag, PVP-coated nano-Ag, nano-Al_2O_3, nano-C60, CNT, nano-Cu, nano-CuO, nano-TiO_2, nano-ZnO and nano-CeO_2 by also considering the particle characteristics. Another recent study constructed a SSD for metallic nanomaterials based on data, that are grouped according to the characteristics of ENMs, test conditions and types of endpoints [59].

In the field of environmental risk assessment, early studies have used the lowest response concentration to evaluate adverse effect of nano-TiO_2, nano-ZnO, nano-Ag, carbon nanotubes, and fullerenes [35, 37]. Later, researchers have also estimated risks posed by ENMs from specific nanoproducts, e.g. a glass cleaner [61, 140, 159]. However, these studies either calculated the PNEC value using the lowest effect concentration or estimated the release of ENMs only from a single application. Other studies have assessed the risks from all possible applications using a PMFA model and a PSSD for nano-TiO_2, nano-Ag, nano-ZnO, fullerenes, CNT, nano-SiO_2 and nano iron oxides for fresh waters, soils, and sediments [55, 56, 172, 189]. In both parts of the risk assessment, the exposure and the hazard assessment, the scientific progress is fast with new exposure models and large numbers of ecotoxicological studies being constantly published that can be used in new risk evaluations.

In light of these issues, the aim of this work was to conduct an environmental risk assessment for ENM that have not or only marginally been covered (nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots). The PEC was modeled using the DPMFA model described in Wang & Nowack (2017) and the hazard assessment is based on updated ecotoxicological data using a PSSD approach [32]. The results from the current study provide insights into the possible risk posed by the considered ENMs in the future based on the predicted release level, and contribute to future regulatory risk assessment framework for ENMs [228].

A4.2 Materials and methods

A4.2.1 Predicted environmental concentration

The exposure assessment for nano-SiO_2, nano iron oxides, nano-CeO_2, nano-Al_2O_3, and quantum dots using the DPMFA model has been described in Wang & Nowack (2017). This study shows that the PEC in fresh water compartments in Europe is the lowest in northern Europe and the highest in southeastern Europe (of seven European regions considered). These two regions were therefore selected in the present work. Due to the increase in production over time, the PEC values are constantly increasing in environmental compartments. In order to consider also the near-future increase, we selected the year 2020 as basis for the model. Therefore, the PEC values in northern Europe and southeastern Europe were extracted from Wang & Nowack (2017) for the year 2020. These PEC values represent average concentrations in standard environmental compartments as defined in the REACH guidance [225].

A4.2.2 Ecotoxicological data collection

Ecotoxicological data for nano-Al_2O_3, nano-SiO_2, nano iron oxides, nano-CeO_2, and quantum dots were collected from peer-reviewed studies published before July 2017. The data screening followed the criteria described in Coll et al (2016). All endpoints from the same study using nanoparticles with different size and/or type (e.g. coating) were considered as individual data points. If tests in one study were conducted under various test conditions, maximal three values were chosen (minimum, median, and maximum). Hence, the collected data covered studies investigating the ecotoxicity of a range of species exposed to varied test materials with different particles sizes and test media. Too little information regarding the hazard of the considered ENMs to soil and sediment organisms was found to build a PSSD for these systems. Hence, the data presented here is restricted to the risk assessment of the targeted ENMs in fresh waters.

The summary of collected ecotoxicological concentrations is given in Table A4.1 and the detailed information extracted from the studies is presented in Table S4.1. In total 173 endpoints were collected, including EC10, EC15, EC20, EC50, LC10 LC20, LC50, IC50, LOEC, HONEC, and NOEC. Among the five ENMs, nano-CeO_2 has the most data available with 71 endpoint concentrations covering 17 species, and the least data is available for quantum dots with 16 data points from 7 species.

Table A4.1. Summary of endpoint concentrations collected for the five considered ENMs

ENM	**Number of species**	**Number of Endpoints**
nano-SiO_2	12	34
nano iron oxides	13	26
nano-Al_2O_3	16	26
nano-CeO_2	17	71
Quantum dots	7	16

A4.2.3 Probabilistic species sensitivity distribution modelling

According to the European Chemical Agency guidelines, the calculation of PNEC values for fresh waters using the SSD method should be based on all available NOECs from long term studies [48]. In this work, two assessment factors (AF) were used to derive chronic NOEC values from the collected data. One AF is used to derive the no observed concentration from the observed concentration if a NOEC is not available. The other AF is applied to extrapolate long-term effects from short-term studies. The method of assigning the AF was described in detail in the previous studies [55, 189]. All input data were used in the model as triangular distributions obtained by multiplying or dividing each value with 2 to obtain a range of final NOEC values that includes the uncertainty of the AF. This method was first developed by Gottschalk and Nowack (2013), and has been applied to generate SSDs in several risk assessment studies [55, 172]. With the Monte Carlo approach in the model, a probabilistic distribution was derived and an overall PSSD of each ENM was then generated by combining all data of single species. The PNEC value was obtained by abstracting the 5th quantile from PSSD. By running the whole calculation 10,000 times, a PNEC distribution was derived (each distribution contains 10,000 random values).

A4.2.4 Risk calculation

The environmental risks of ENMs were quantified by calculating the risk characterization ratio (RCR) by comparing the exposure level (PEC) with the corresponding PNEC according to the risk assessment guidelines (Equation 1) [225]. If the RCR is less than 1, the risk to the environment is controlled at the given exposure scenario. If the RCR is larger than 1, certain risk management measures have to be taken. The RCR distribution was generated by dividing each values of the PEC distribution by all data from the PNEC distribution of each ENM.

$$\mathrm{RCR} = \mathrm{PEC} / \mathrm{PNEC} \qquad (1)$$

A4.3 Results

The PEC values for fresh waters are based on the dynamic MFA model by Wang and Nowack (2017) and therefore consider the accumulation of ENM in stocks and the delayed release from them. The dynamic nature of the model also allows an extrapolation of the release into the future and 2020 has been chosen as reference year to enable a precautionary assessment. The mean concentrations of the five considered ENMs in 2020 in northern Europe and southeastern Europe, as well as the 5th and 95th percentiles of the distribution, are given in Table A4.2. Quantum dots have the lowest concentration among the five studied ENMs at the level of fg/l and the highest concentrations are observed for nano-SiO_2 at the µg/l level. Based on the study by Wang and Nowack (2017), northern and southeastern Europe were chosen because they represent the European regions with the lowest and highest fresh water concentrations of ENM, mainly caused by their different volume of fresh waters and the different level of wastewater treatment infrastructure.

Table A4.2. Predicted environmental concentrations of five ENMs in the fresh water in 2020 based on the model by Wang and Nowack (2017)

ENM	Northern Europe	Southeastern Europe	Unit
nano-SiO_2	562 (5.35 - 1640)	2600 (21.7 - 8460)	ng/l
nano iron oxides	12.8 (0.271 - 55.5)	44.2 (0.872 - 171)	ng/l
nano-Al_2O_3	39.6 (0.716 - 99)	221 (3.48 - 780)	ng/l
nano-CeO_2	268 (5.81 - 1230)	1130 (23.2 - 5430)	pg/l
Quantum dots	32.8 (10.9- 89.3)	107 (14.9 - 369)	fg/l

Note: Means values are given, and the values in the bracket are the 5th and 95th percentiles.

The NOEC values calculated from the reported endpoints are contained in Table S4.1 and have been used to construct the PSSD for each material shown in Figure A4.1. The NOEC values of individual species are indicated with blue triangles, the calculated PSSD with the red line. The NOEC values for one species exposed to the same ENM from different studies can vary a lot. For example, the highest NOEC value of *P. subcapita* for nano-CeO_2 is five orders of magnitude higher than the lowest value. The range of response concentrations between the most and the least sensitive species for all considered ENMs is between four and six orders of magnitude.

The PNEC values, which are derived from the 5th percentile of the PSSD, are given in Table A4.3. The highest PNEC is found for nano-SiO_2 with 220 µg/l, the lowest one for quantum dots with 0.32 µg/l. Figure A4.2 shows the comparision of whole PNEC distribution (red line) and the corresponding PEC distributions for each of the ENM in the fresh water compartment in northern Europe (blue line) and in southeastern Europe (green line) in 2020. The PEC distribtution of southeastern Europe is found to be closer to the PNEC compared to the PEC of northern Europe. This is because the concentration of ENMs in 2020 in southeastern Europe is higher than that in northern Europe. Overall, no overlap was observed between PEC and PNEC distributions for any of the considered ENMs. However, the distance between the PEC and PNEC curves of these ENMs varies a lot. For example, for nano-$SiO_{2,}$ the PEC and PNEC curves are quite close to each other, while the gap for quantum dots is much larger. To quantify this gap, we calculated the RCR.

Table A4.3. Mean values of predicted no effect concentrations (PNEC) and risk characteristic ratio (RCR)

ENM	PNEC (µg/l)	RCR (NE)	RCR (SEE)
nano-SiO_2	220 (196 - 248)	2.21E-03 (2.25E-05 - 7.16E-03)	1.13E-02 (9.85E-05 - 3.86E-02)
nano iron oxides	128 (84 - 169)	1.10E-04 (2.18E-06 - 6.27E-04)	3.64E-04 (8.25E-06 - 1.67E-03)
nano-Al_2O_3	120 (107 - 134)	3.21E-04 (6.68E-06 - 8.57E-04)	1.93E-03 (3.13E-05 - 6.41E-03)
nano-CeO_2	2.6 (2.1 - 3.2)	9.14E-05 (2.33E-06 - 4.09E-04)	3.6E-04 (7.88E-06 - 1.58E-03)
Quantum dots	0.32 (0.07 - 1.11)	2.43E-07 (1.75E-08 - 8.06E-07)	7.84E-07 (3.59E-08 - 3.13E-06)

Note: the values in the bracket are the 5th and 95th percentiles. NE: Northern Europe, SEE: Southeastern Europe

The mean RCR values and the range between the 5th and 95th percentiles are also given in Table 5-3. The complete probability distributions of the five RCR are shown in Figure A4.3. All RCR values are at three to seven orders of magnitudes lower than 1, and the order of RCR values among all ENMs is nano-SiO_2 > nano-Al_2O_3 > nano iron oxides > nano-CeO_2 > quantum dots. The risk posed by the considered ENMs in 2020 in northern Europe is smaller than that in southeastern Europe.

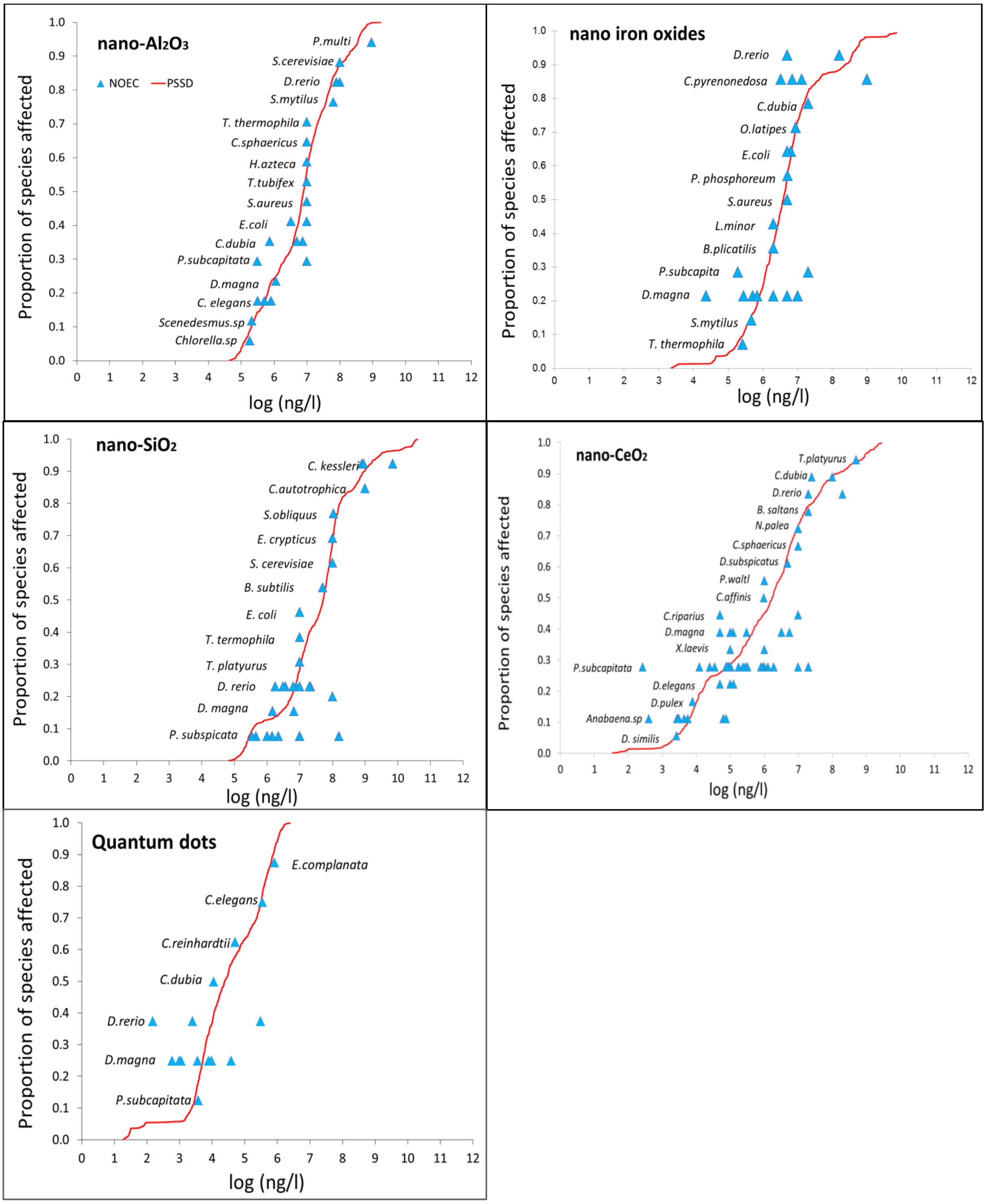


Figure A4.1. Probabilistic species sensitivity distribution (PSSD) for nano-Al_2O_3, nano-SiO_2, nano iron oxides, nano-CeO_2, and quantum dots in fresh water. The blue triangles represent no observed effect concentration (NOEC), which are calculated based on collected endpoint concentrations. The red lines are the PSSD.

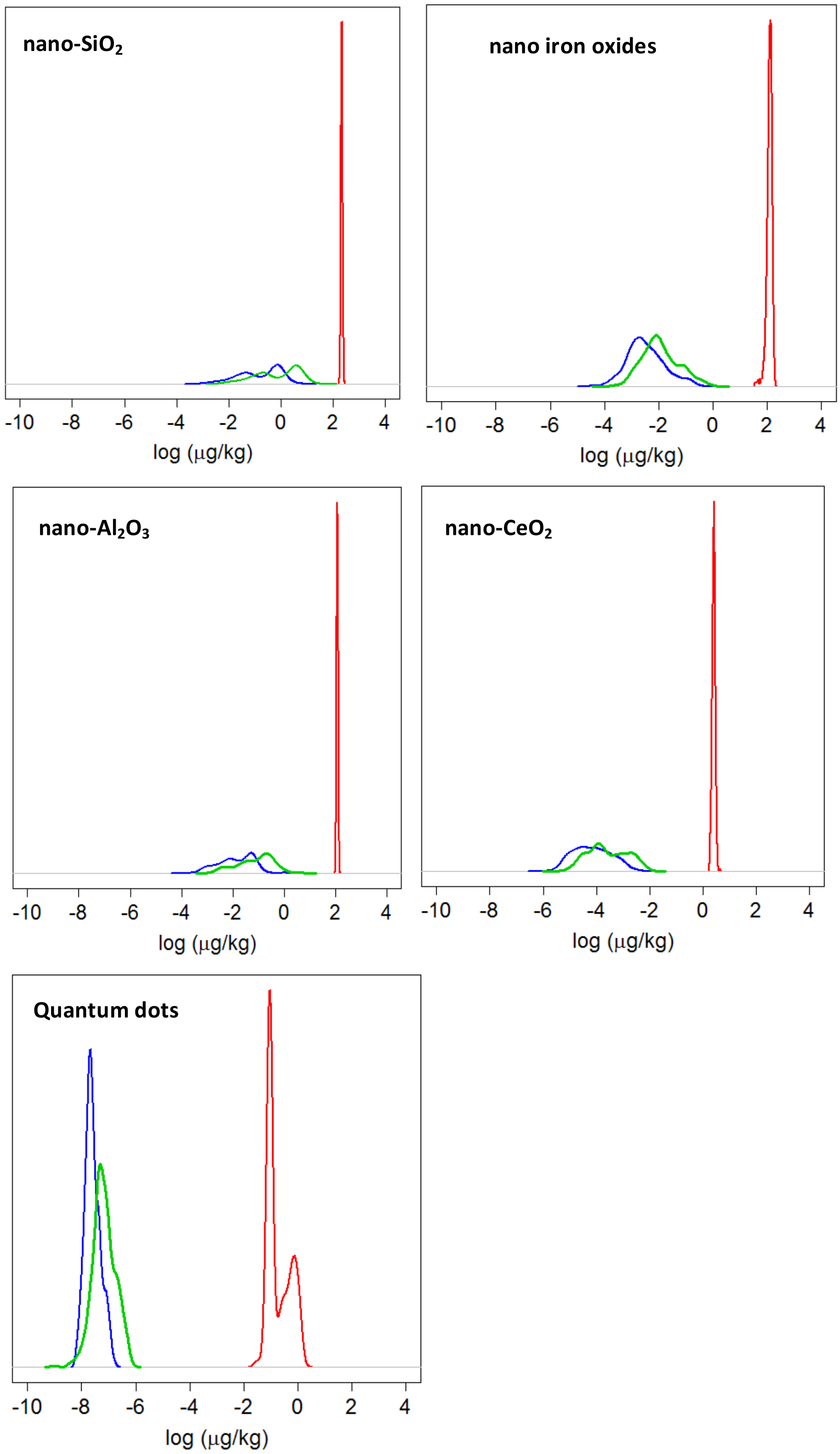


Figure A4.2. Comparison of predicted no-effect concentration (PNEC) (red line) with predicted environmental concentrations (PEC) of nano-Al_2O_3*, nano-*SiO_2*, nano iron oxides, nano-*CeO_2*, and quantum dots in fresh water in northern Europe (blue line) and in southeastern Europe (green line) in 2020.*

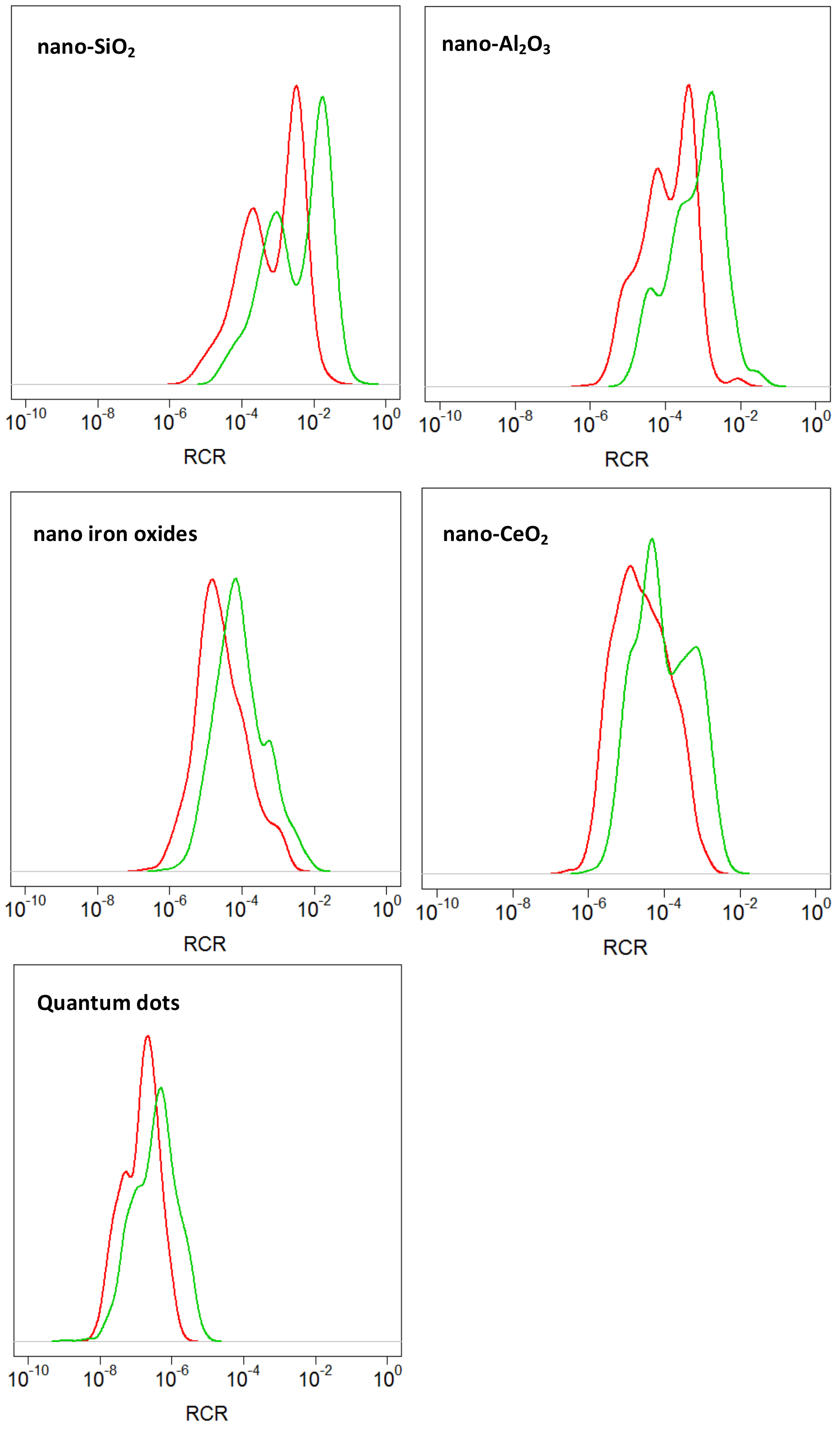


Figure A4.3. Risk characterization ratio (RCR) distribution of nano-Al_2O_3*, nano-*SiO_2*, nano iron oxides, nano-*CeO_2*, and quantum dots in northern Europe (red line) and in southeastern Europe (green line) in 2020.*

A4.4. Discussion

This study is the first to conduct an environmental risk assessment for ENM using release data based on dynamic material flow modeling. This dynamic model is considering a material input into the environment from 1990 to 2020 and also reflects regional differences in waste management [227]. From previous static models, PEC values are available for nano-SiO_2, nano iron oxides, nano-Al_2O_3, nano-CeO_2, and quantum dots [42, 172, 189]. The concentrations presented in the current paper can be either higher or lower than those from previous studies, depending on the material. This can be explained by the following factors: i) the different input data and methods used to assess production volume and product allocation that are at the basis for the different models; ii) the various time periods or geographic systems considered in the studies; iii) the different modelling approaches used to predict PEC and PNEC. For example, the dynamic model considers both the increasing production of ENM over time as well as time-dependent releases and accumulation of ENMs in stocks and environmental compartments. Therefore, the PEC values in 2020 in this work are slightly larger than that predicted in the previous study using the same DPMFA model as the production and use of ENMs have been increasing after 2014, the base year in the previous model [227].

The DPMFA model allows us to predict environmental concentrations considering the changing behavior of the model system over time. However, it does not include one fact, which is the change of nanoapplications over time. Due to an almost complete lack of data on historic product distributions for ENM, the model is based on the simplifying assumption that the product distribution in the past was the same as currently and that it remains the same in the near future. Sun et al. (2017) have used the dynamic MFA model to predict future scenarios of changing product distributions which could be used in the future in prospective risk assessments.

It is necessary to state that the risk assessment in the current study is generic for each modelled ENM, namely that the exposure and hazard assessments are not specific in terms of type of ENM. The PNEC values derived in the current study did not distinguish ENMs with various sizes, forms, and test conditions. However, researchers have demonstrated that the type of crystal phase can change the toxic effects of ENMs significantly [229]. A first study by Gottschalk et al (2015) provides PEC values for different forms of nano-TiO_2, which could form the basis to assess the environmental risk posed by the different form of nano-TiO_2 (e.g. photocatalytic and photostable) [42].

The exposure assessment in this study estimated the total flux and concentrations of each considered ENM in the environment without considering the transformation of ENMs. Therefore, the exposure level predicted for 2020 is rather conservative under a worst scenario with no transformations included. ENMs are affected in the environment by a series of reactions such as agglomeration, dissolution, and sedimentation [43, 230]. These reactions will decrease the exposure concentration in water and thus decrease the calculated risks even further.

Another aspect determining the PEC values that needs consideration is the fact that the predicted concentration is represents an average level for the target region. One of the possible release pathways of ENM entry to the environment is from production and manufacturing, with the effluent of sewage treatment plants and by emissions from waste incineration plants. These releases are local point sources and their geographic distribution determines the local PEC values that can vary greatly between different regions [40, 89, 90, 194, 214]. For example, Dumont et al (2015) modelled nano-Ag and nano-ZnO monthly concentrations in European fresh waters by considering dilution, downstream transport, water evaporation, water abstraction, and nano-particle sedimentation. High concentrations of ENM were predicted close to big cities due to the high population density and thus high wastewater production. The concentrations estimated in our work are valid for the regional scale, which implies that the PEC values in a local water system might be higher than the predicted values and consequently may underestimate the environmental risk of the considered ENMs.

Earlier studies have already applied the SSD method to assess the environmental hazards for some of the ENM covered in our study, e.g. nano-SiO_2, nano iron oxides, nano-Al_2O_3, and nano-CeO_2 [57, 172, 189]. The PNEC value of nano-SiO_2 predicted in the current study is five times smaller than in [172], caused by new ecotoxicological studies reporting effects at low concentrations. The updated PNEC for nano iron oxides is at the same concentration level than in a previous study [189]. Garner et al. (2015) have built SSDs for nano-Al_2O_3, and nano-CeO_2 as well and estimated that the PNEC are 2-8 mg/l for nano-Al_2O_3, and 0.08-6 mg/l for nano-CeO_2 [57]. The PNEC range calculated in our work are 107-134 µg/l (nano-Al_2O_3) and 2.1-3.2 µg/l (nano-CeO_2), which is one to three orders of magnitudes lower compared to the previous study. The reason for this difference is that the SSD built by Garner et al. (2015) was for acute freshwater toxicity using LC50 values whereas we used two different assessment factors to convert acute observe effect concentration to chronic NOEC values, thus explaining up to a factor of 100 of difference between these two studies.

A handful of studies have investigated the influence of ENM characteristics on the effects of ENM [57, 59, 159, 231]. Garner et al. (2015) for example built SSDs for metal nanomaterials separated into different forms, e.g. uncoated Ag, Ag-PVP, and Ag+, and only minor differences were observed between coated Ag particles and Ag+, and the uncoated Ag was found to be less toxic than PVP coated Ag. Other factors including size, exposure time, and shape of the nanoparticles were also shown to influence the toxic effects of ENMs [59]. However, according to the REACH guidelines, building an SSD requires at least ten endpoint concentrations from different species for at least eight taxonomic groups [225]. If enough data are available for constructing more detailed SSDs, the hazard assessment of the current work could be improved with considering the effect of characterization of ENMs and the experimental conditions. However, at the same time exposure data for the different forms of the ENM would also need to be available, which is currently not the case.

Our risk assessment shows no overlap between the PEC distributions in fresh waters and PNEC distributions in all cases. Therefore, no risk is predicted under the estimated release level in 2020 for the five considered ENMs. RCR values are decided by both the exposure and the effect level and therefore the ENM with the highest hazard to organisms does not necessarily imply the highest associated risk. For example, quantum dots show the smallest risk even though they are the most toxic ENMs among all considered ENMs. Nano-SiO_2 is the ENM of the highest concerns with the largest RCR value although it has the highest PNEC value. This is because the exposure concentration of nano-SiO_2 is several orders of magnitude higher than that of quantum dots – it is the ratio of the two values that matters for the risk.

Thousands of nano-ecotoxicological papers have been published since 2004, however, the data is still inadequate to conduct a full risk assessment for every environmental compartment [49]. The soil and sediment compartments were not included in the current study due to a limited number of ecotoxicological studies reporting EC50 or NOEC values to construct an SSD. However, sufficient evidence has proven that ENMs are likely to accumulate in soils and sediments because these compartments constitute sinks for these materials and critical ENM concentrations might be reached over time [43, 44, 232]. Because our PEC modeling also provides values for soils and sediments, the risks to the compartments can easily be calculated once enough ecotoxicological data become available to build an SSD.

The results from this study can provide regulators scientific foundations for a risk-based environmental policy regarding ENMs in the future. This work will complement the growing number of risk assessment frameworks that are proposed or under development to deal with the specific challenges existing in the risk assessment process for ENM [24, 228, 233].

A4.5 References

[1] Schmid K, Riediker M. 2008. Use of nanoparticles in Swiss industry: A targeted survey. *Environ Sci Technol* 42:2253-2260.

[2] Zhang YY, Leu YR, Aitken RJ, Riediker M. 2015. Inventory of Engineered Nanoparticle-Containing Consumer Products Available in the Singapore Retail Market and Likelihood of Release into the Aquatic Environment. *International journal of environmental research and public health* 12:8717-8743.

[3] Wiesner MR, Lowry GV, Alvarez P, Dionysiou D, Biswas P. 2006. Assessing the risks of manufactured nanomaterials. *Environ Sci Technol* 40:4336-4345.

[4] Barlow PG, Donaldson K, MacCallum J, Clouter A, Stone V. 2005. Serum exposed to nanoparticle carbon black displays increased potential to induce macrophage migration. *Toxicol Lett* 155:397-401.

[5] Handy RD, van der Kammer F, Lead JR, Hassellöv M, Owen R, Crane M. 2008. The ecotoxicology and chemistry of manufactured nanoparticles. *Ecotoxicology* 17:287-314.

[6] Hansen S, Larsen B, Olsen S, Baun A. 2007. Categorization framework to aid hazard identification for nanomaterials. . *Nanotoxicology* 1:243-250.

[7] ECHA. 2016. Guidance on information requirements and Chemical Safety Assessment
Chapter R.16: Environmental exposure.

[8] von der Kammer F, Ferguson PL, Holden PA, Masion A, Rogers KR, Klaine SJ, Koelmans AA, Horne N, Unrine JM. 2012. Analysis of engineered nanomaterials in complex matrices (environment and biota): General considerations and conceptual case studies. *Environ Toxicol Chem* 31:32-49.

[9] Nowack B, Baalousha M, Bornhoft N, Chaudhry Q, Cornelis G, Cotterill J, Gondikas A, Hassellov M, Lead J, Mitrano DM, von der Kammer F, Wontner-Smith T. 2015. Progress towards the validation of modeled environmental concentrations of engineered nanomaterials by analytical measurements. *Environ Sci-Nano* 2:421-428.

[10] Hendren CO, Lowry M, Grieger KD, Money ES, Johnston JM, Wiesner MR, Beaulieu SM. 2013. Modeling Approaches for Characterizing and Evaluating Environmental Exposure to Engineered Nanomaterials in Support of Risk-Based Decision Making. *Environmental Science & Technology* 47:1190-1205.

[11] Baalousha M, Cornelis G, Kuhlbusch TAJ, Lynch I, Nickel C, Peijnenburg W, van den Brink NW. 2016. Modeling nanomaterial fate and uptake in the environment: current knowledge and future trends. *Environ Sci-Nano* 3:323-345.

[12] Nowack B. 2017. Evaluation of environmental exposure models for engineered nanomaterials in a regulatory context. *NanoImpact* 8:38-47.

[13] Bornhoft NA, Sun TY, Hilty LM, Nowack B. 2016. A dynamic probabilistic material flow modeling method. *Environ Modell Softw* 76:69-80.

[14] Sun TY, Bornhoft NA, Hungerbuhler K, Nowack B. 2016. Dynamic Probabilistic Modeling of Environmental Emissions of Engineered Nanomaterials. *Environ Sci Technol* 50:4701-4711.

[15] Sun TY, Mitrano DM, Bornhöft NA, Scheringer M, Hungerbühler K, Nowack B. 2017. Envisioning Nano Release Dynamics in a Changing World: Using Dynamic Probabilistic Modeling to Assess Future Environmental Emissions of Engineered Nanomaterials. *Environmental Science & Technology* 51:10.

[16] Wang Y, Nowack B. 2017. Dynamic probabilistic material flow analysis of nano-SiO2, nano iron oxides, nano-CeO2, nano-Al2O3, and quantum dots in seven European regions (submitted). *Environmental pollution*.

[17] Fox DR. 2010. A Bayesian approach for determining the no effect concentration and hazardous concentration in ecotoxicology. *Ecotoxicology and environmental safety* 73:123-131.

[18] Frampton GK, Jansch S, Scott-Fordsmand JJ, Rombke J, Van den Brink PJ. 2006. Effects of pesticides on soil invertebrates in laboratory studies: a review and analysis using species sensitivity distributions. *Environ Toxicol Chem* 25:2480-2489.

[19] Kooijman SALM. 1987. A Safety Factor for Lc50 Values Allowing for Differences in Sensitivity among Species. *Water Res* 21:269-276.

[20] Gottschalk F, Nowack B. 2012. A probabilistic method for species sensitivity distributions taking into account the inherent uncertainty and variability of effects to estimate environmental risk. *Integrated Environmental Assessment and Management* 9:79–86.

[21] Coll C, Notter D, Gottschalk F, Sun T, Som C, Nowack B. 2016. Probabilistic environmental risk assessment of five nanomaterials (nano-TiO_2, nano-Ag, nano-ZnO, CNT, and fullerenes). *Nanotoxicology* 10:436-444.

[22] Semenzin E, Lanzellotto E, Hristozov D, Critto A, Zabeo A, Giubilato E, Marcomini A. 2015. Species sensitivity weighted distribution for ecological risk assessment of engineered nanomaterials: the n-TiO2 case study. *Environ Toxicol Chem* 34:2644-2659.

[23] Gottschalk F, Kost E, Nowack B. 2013. Engineered nanomaterials (ENM) in waters and soils: a risk quantification based on probabilistic exposure and effect modeling. *Environ Toxicol Chem* 32:1278-1287.

[24] Garner KL, Suh S, Lenihan HS, Keller AA. 2015. Species sensitivity distributions for engineered nanomaterials. *Environ Sci Technol* 49:5753-5759.

[25] Chen G, Peijnenburg WJ, Xiao Y, Vijver MG. 2017. Developing species sensitivity distributions for metallic nanomaterials considering the characteristics of nanomaterials, experimental conditions, and different types of endpoints. *Food Chem Toxicol*.

[26] Mueller NC, Nowack B. 2008. Exposure modeling of engineered nanoparticles in the environment. *Environ Sci Technol* 42:4447-4453.

[27] Gottschalk F, Sonderer T, Scholz RW, Nowack B. 2009. Modeled environmental concentrations of engineered nanomaterials (TiO_2, ZnO, Ag, CNT, fullerenes) for different regions. *Environ Sci Technol* 43:9216-9222.

[28] Michel K, Scheel J, Karsten S, Stelter N, Wind T. 2013. Risk assessment of amorphous silicon dioxide nanoparticles in a glass cleaner formulation. *Nanotoxicology* 7:974-988.

[29] Dekkers S, Krystek P, Peters RJB, Lankveld DPK, Bokkers BGH, van Hoeven-Arentzen PH, Bouwmeester H, Oomen AG. 2011. Presence and risks of nanosilica in food products. *Nanotoxicology* 5:393-405.

[30] Mahapatra I, Sun TY, Clark JRA, Dobson PJ, Hungerbuehler K, Owen R, Nowack B, Lead J. 2015. Probabilistic modelling of prospective environmental concentrations of gold nanoparticles from medical applications as a basis for risk assessment. *Journal of nanobiotechnology* 13:1-14.

[31] Wang Y, Kalinina A, Sun T, Nowack B. 2016. Probabilistic modeling of the flows and environmental risks of nano-silica. *Science of the Total Environment* 545-546:67-76.

[32] Wang Y, Deng L, Caballero-Guzman A, Nowack B. 2016. Are engineered nano iron oxide particles safe? an environmental risk assessment by probabilistic exposure, effects and risk modeling. *Nanotoxicology* 10:1545-1554.

[33] Steinhäuser KS, Sayre PG. 2017. Reliability of methods and data for regulatory assessment of nanomaterial risks. *NanoImpact* 7:66-74.

[34] ECHA. 2008. Guidance on information requirements and chemical safety assessment. Chapter R.10: Characterisation of dose [concentration]-response for environment.

[35] Gottschalk F, Lassen C, Kjoelholt J, Christensen F, Nowack B. 2015. Modeling flows and concentrations of nine engineered nanomaterials in the Danish environment. *International journal of environmental research and public health* 12:5581-5602.

[36] Lin X, Li J, Ma S, Liu G, Yang K, Tong M, Lin D. 2014. Toxicity of TiO2 nanoparticles to Escherichia coli: effects of particle size, crystal phase and water chemistry. *Plos One* 9:e110247.

[37] Klaine SJ, Alvarez PJJ, Batley GE, Fernandes TF, Handy RD, Lyon DY, Mahendra S, McLaughlin MJ, Lead JR. 2008. Nanomaterials in the environment: Behavior, fate, bioavailability, and effects. *Environ Toxicol Chem* 27:1825-1851.

[38] Baun A, Sayre P, Steinhäuser KG, Rose J. 2017. Regulatory relevant and reliable methods and data for determining the environmental fate of manufactured nanomaterials. *NanoImpact* 8:1-10.

[39] Gottschalk F, Ort C, Scholz RW, Nowack B. 2011. Engineered nanomaterials in rivers - exposure scenarios for Switzerland at high spatial and temporal resolution *Environ Pollut* 159:3439-3445.

[40] Sun TY, Conroy G, Donner E, Hungerbuhler K, Lombi E, Nowack B. 2015. Probabilistic modelling of engineered nanomaterial emissions to the environment: a spatio-temporal approach. *Environ Sci-Nano* 2:340-351.

[41] Dumont E, Johnson AC, Keller VDJ, Williams RJ. 2015. Nano silver and nano zinc-oxide in surface waters - Exposure estimation for Europe at high spatial and temporal resolution. *Environ Pollut* 196:341-349.

[42] Keller AA, Lazareva A. 2014. Predicted Releases of Engineered Nanomaterials: From Global to Regional to Local. *Environ Sci Tech Let* 1:65-70.

[43] Dale AL, Lowry GV, Casman EA. 2015. Stream Dynamics and Chemical Transformations Control the Environmental Fate of Silver and Zinc Oxide Nanoparticles in a Watershed-Scale Model. *Environmental Science & Technology* 49:7285-7293.

[44] Botha TL, James TE, Wepener V. 2015. Comparative Aquatic Toxicity of Gold Nanoparticles and Ionic Gold Using a Species Sensitivity Distribution Approach. *J Nanomater* 2015:1-16.

[45] Hjorth R, Skjolding LM, Sorensen SN, Baun A. 2017. Regulatory adequacy of aquatic ecotoxicity testing of nanomaterials. *NanoImpact* 8:28-37.

[46] Pan B, Xing BS. 2012. Applications and implications of manufactured nanoparticles in soils: a review. *European Journal of Soil Science* 63:437-456.

[47] Gardea-Torresdey JL, Rico CM, White JC. 2014. Trophic Transfer, Transformation, and Impact of Engineered Nanomaterials in Terrestrial Environments. *Environmental Science & Technology* 48:2526-2540.

[48] Sayre PG, Steinhäuser KG, van Teunenbroek T. 2017. Methods and data for regulatory risk assessment of nanomaterials: Questions for an expert consultation. *NanoImpact* 8:20-27.

[49] Hristozov D, Gottardo S, Semenzin E, Oomen A, Bos P, Peijnenburg W, Van Tongeren M, Nowack B, Hunt N, Brunelli A, Scott-Fordsmand J, Marcomini A. 2016. Frameworks and tools for Risk Assessment and Management of Manufactured Nanomaterials. *Environ Int* 95:36-53.

Supporting information of Annex 1

Estimates of production of nano-silica

Table S1.1. Data sources for the production volume of nano-silica in Europe and Switzerland with degree of belief

Worldwide [t/a]	**EU [t/a]**	**Switzerland [t/a]**
Class 1: (80% DoB)		
185,000-1,400,000 [1]	32,448-245,549	824-6,239
	5,500 [2]	140
18,000 [3]	3,157	80
	2,965	75.4[4]
Class 2: (20% DoB)		
	836,467 [5]	25,172
	200-1,000 [6]	5-25
100,000 [7]	17,539	446
100-1000 [8]	17.5-175	0.446-4.46
481,050 [9]	84,372	2,143
1,000,000 [10]	175,392	4,456

Table S1.2 Global and regional PPP GDP and scaling factors

According to the GDP ppp 2014 (US dollar)					
World	EU	Switzerland	US	France	
1.02573E+14	1.79901E+13	4.57129E+11	1.67681E+13	2.47488E+12	
Scaling factors					
EU-CH	World-CH	World-EU	EU-US	US-CH	EU-FR
39.329	224.385	5.7015	1.073	36.681	7.2691

Allocation of nano-silica to product categories

Five different inventories and sources were checked to define the product categories. As shown in Table S3, twenty product categories were collected and the share of each was also summarized. The minimum, mean and maximum values of the fraction of each category were determined and then used for modeling triangle distributions.

Table S1.3 Product categories and the share of each in the total production volume

Own categories									
Items	**Product Categories**	**WWI (2011)**	**BUND (2011)**	**Recent Patents on Nanosilica**	**EC21 (2011)**	**The Nano database**	**Min**	**Mean**	**Max**
1	Paints	35.9220	85.2165	29.2404	63.3761	99.8970	0.2924	0.6273	0.9990
2	Polymers	0.0000	0.0000	30.5452	5.0926	0.0000	0.0000	0.0713	0.3055
3	Coatings	15.9653	3.7874	14.2953	1.4084	0.0000	0.0000	0.0709	0.1597
4	Stone, cement, glass and ceramics	0.0000	1.0605	11.8261	19.7170	0.0000	0.0000	0.0652	0.1972
5	Car tires	28.7376	0.0000	0.0000	0.0000	0.0000	0.0000	0.0575	0.2874
6	Cleaning agents	17.8029	1.2067	2.0702	4.4870	0.0000	0.0000	0.0511	0.1780
7	Catalysts	0.0000	0.0000	9.7468	4.2251	0.0000	0.0000	0.0279	0.0975
8	Food products	0.0000	7.2112	0.0000	0.0000	0.0000	0.0000	0.0144	0.0721
9	Cosmetics, personal care products	0.9579	1.4771	0.1949	1.4788	0.0000	0.0000	0.0082	0.0148
10	**Others**								
11	Adhesive and sealants	0.4445	0.0000	0.4221	0.0000	0.1030	0.0000	0.0019	0.0044
12	Composites and alloys	0.0000	0.0000	0.8296	0.0000	0.0000	0.0000	0.0017	0.0083
13	Machinery, mechanical appliances	0.0000	0.0000	0.1825	0.1977	0.0000	0.0000	0.0008	0.0020
14	Inks and toners	0.0000	0.0000	0.3680	0.0000	0.0000	0.0000	0.0007	0.0037
15	Lubricants, greases, release agents	0.0000	0.0000	0.1326	0.0000	0.0000	0.0000	0.0003	0.0013
16	Dental applications	0.0512	0.0000	0.0625	0.0000	0.0000	0.0000	0.0002	0.0006
17	Sporting goods	0.0525	0.0100	0.0000	0.0000	0.0000	0.0000	0.0001	0.0005
18	Articles for health services	0.0275	0.0174	0.0075	0.0081	0.0000	0.0000	0.0001	0.0003
19	Putties, plasters, modeling clay	0.0000	0.0000	0.0546	0.0000	0.0000	0.0000	0.0001	0.0005
20	Clothes and shoes	0.0281	0.0133	0.0114	0.0000	0.0000	0.0000	0.0001	0.0003
21	Electrical / electronic products	0.0105	0.0000	0.0102	0.0092	0.0000	0.0000	0.0001	0.0001

Keller's categories		
Items	**Product Categories**	**Mean**
1	Automotive	20.0000
2	Electronics & Optics	20.0000
3	Energy & Environment	20.0000
4	Coating, Paints & Pigments	10.0000
5	Catalysts	20.0000
6	Sensors	10.0000

Transfer coefficients

The transfer coefficients for nano-silica were collected through a literature review and estimated based on the current knowledge about relevant processes. Table S4 shows the transfer coefficients of nano-silica from two categorizations (own category and Keller's category) to environmental and technical compartments. The release of nano-silica from production and manufacturing processes to the environment is negligible (assumed from 0 to 2 %), due to the closed production or manufacturing system and solvent free procedures [11]. Some product categories have no release to the environment such as the category “Inks and toner”, and almost every kind product is disposed to landfill except the category “Lubricants, greases, release agents” and “Catalysts” from the “own” categorization and the category “Automotive” from Keller's categorization. The main release of nano-silica from the category “Car tires” is to the soil, recycling, export compartments and to cement industry, where tires can be used as an alternative fuel. According to European regulations, car tires are not allowed to be landfilled since hazardous substances are generated from the rubber polymers. Nano-silica, apart from the PMC phases, is entering sediments only through sedimentation from surface water, where nano-silica comes from wastewater unconnected to a STP, STP effluent, Air and Soil.

Information regarding the sewage treatment plant (STP) removal efficiency was collected from current available experimental studies and a DoB was assigned to each data source (shown in Table S5). A recent study showed that the removal efficiency in wastewater treatment plant is more than 97% [12]. In another study it was concluded the surface functionalized silica nanoparticles are likely to be removed during the primary treatment process and the removal rate is up to 94% [13]. Other earlier studies, to which a low DoB was assigned, used turbidity as a monitoring parameter to assess the efficiency of treatment process and it showed up to 99% of the turbidity can be removed from aggregation [8, 14, 15]. The STP removal efficiency of nano-silica was modelled and illustrated in the form of probability density distribution using the current probabilistic approach. The STP removal efficiency distribution is presented in Figure S1.

Table S1.4. Transfer coefficients from PMC and products to environmental and technical compartments

Product category	Waste water	WIP	Landfill	Air	Soil	Surface water	Cement	Recycling	Export
Own categories									
Production	0.75			0.25					
Manufacturing	0.33	0.11	0.22	0.34					
Polymers		0.32	0.68						
Electrical / electronic products		0.05	0.1	0.03	0.02			0.65	0.2
Machinery, mechanical appliances	0.05	0.02	0.03					0.9	
Stone, cement, glass and ceramics	0.01	0.15	0.25					0.59	
Adhesive and sealants	0.2	0.75	0.025			0.025			
Coatings	0.05	0.02	0.03					0.9	
Inks and toners		0.07	0.21					0.72	
Lubricants, greases, release agents		0.2						0.8	
Articles for health services	0.9	0.05	0.05						
Cleaning agents	0.95	0.025	0.025						
Cosmetics, personal care products	0.85	0.02	0.03			0.1			
Composites and alloys		0.32	0.67	0.01					
Putties, plasters, modeling clay	0.05		0.4		0.1	0.05		0.4	
Paints	0.05		0.4		0.1	0.05		0.4	
Clothes and shoes	0.02-0.32-0.65	Rest*0.4	Rest*0.1					Rest*0.1	Rest*0.4
Catalysts	0.05							0.95	
Car tires		0.037			0.11	0.001	0.4	0.172	0.28
Sporting goods	0.02	0.3	0.66	0.02					
Food products	0.9	0.05	0.05						
Dental applications	0	0.8	0.2						
Keller's categories									
Coating & Paints & Pigments	0.05		0.4		0.1	0.05		0.4	
Sensors		0.005	0.005					0.99	
Automotive		0.037			0.11	0.001	0.4	0.172	0.28
Electronics & Optics		0.25	0.25					0.95	
Energy & Environment		0.005	0.005					0.99	
Catalysts	0.001	0.002	0.002					0.99	

Table S1.5. Data source for sewage treatment plant removal efficiency with Degree of Belief (DoB)

Removal efficiency [%]	DoB	Comments	
94	80	Coated silica nanoparticles	[13]
97	80	DNA encapsulated silica nanoparticles	[12]
81.8 - 92.7	20	Recovery of pure water flux, depending on treatment conditions	[16]
99	20	Turbidity removal	[8]
90	20	Turbidity removal	[14]
98.8	20	Turbidity removal efficiency for electrocoagulation	[17]
96 - 99.3	20	Turbidity removal efficiency for chemical coagulation	[17]
94.5 - 96.1	20	Silica removal	[15]

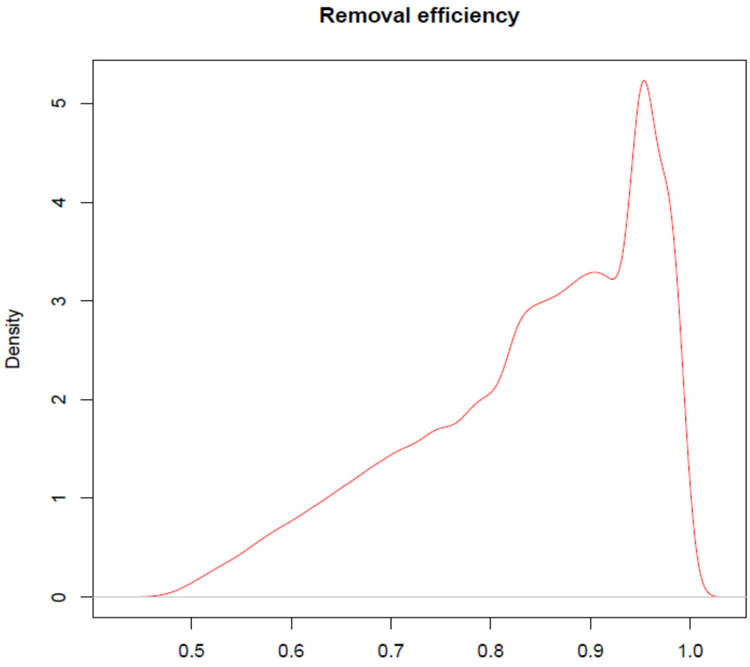


Figure S1.1 Modeled probability distribution of STP removal efficiency of nano-silica

Material flows

Table S1.6. Flows from PMC to the technical or environmental compartments and from the technical compartments to the environmental compartments

	EU					**Switzerland**				
	Unit (tons)									
Flows	**Q15**	**Q85**	**median**	**mode**	**mean**	**Q15**	**Q85**	**median**	**mode**	**mean**
PMC to Wastewater	361	22840	863	783	13131	9.23	586	22	9.38	363
PMC to Landfill	768	47943	1752	1391	26643	20	1225	44	25	743
PMC to WIP	212	13230	495	541	7520	5.33	340	12	4.45	209
PMC to Recycling	926	55415	2008	1435	31010	24	1429	51	40	859
PMC to Export	55	4470	182	89	2636	1.4	116	4.6	4.3	73
PMC to Air	0	554	18	11	303	0	14	0.46	0.32	8.49
PMC to Soil	99	7746	308	328	4426	2.50	200	7.71	4.34	124
PMC to Surface water	5.1	3630	131	100	2076	0	94	3.30	4.18	58
PMC to Cement	79	6539	258	127	3743	2.02	163	6.45	4.65	103
WasteWater to STP	288	18265	691	702	10502	8.99	571	21	8.71	353
Wastewater to Surface water	68	4499	68	156	2629	0.24	15	0.58	0	10
Bottom ash to Recycling	89	2683	216	70	3263	0	0	0	0	0
Bottom ash to Landfill	127	7924	297	253	4508	9.43	578	21	13	349
Filter ash to Air	0.06	4.7	0.19	0.06	2.8	0	0	0	0	0
Filter ash to Landfill	52	3299	126	22	1892	1.57	102	2.00	3.70	33
Filter ash to Export	0.0	0.0	0.0	0.0	0.0	0.48	30	1.12	0.61	19
Filter ash to Acid washing 2	0	0	0	0	0	0.84	54	1.99	1.08	33
Acid washing 1 to Air	0	0.048	0	0	0.003	0	3.94E-04	1.60E-05	0	2.55E-04
Acid washing 1 to Landfill	0.06	4.9	0.20	0.06	2.9	0.01	0.40	0.02	0.01	0.26
Overflows	9.0	583	22	15	335	0.3	18.2	0.68	0.51	11.3
Effluent of STP	18	2476	114	38	1653	0.51	74	3.36	2.53	53
Sludge to Landfill	43	2935	115	20	1717	0	0	0	0	0
Sludge to WIP	54	3646	144	29	2148	1.57	102	3.85	3.70	64
Sludge to Cement	--	--	--	--	--	5.56	361	14	13	226
Sludge to Soil	122	8001	309	278	4648	0	0	0	0	0
Air to Surfacewater	0.0	17	0.6	0.3	9.2	0.00	0.60	0.02	0.01	0.36
Air to Soil	0.45	542	18	11	297	0.00	14	0.44	0.30	8.13
Soil to Surface water	1.4	90	3.4	0.5	51	0.01	1.16	0.04	0.02	0.72

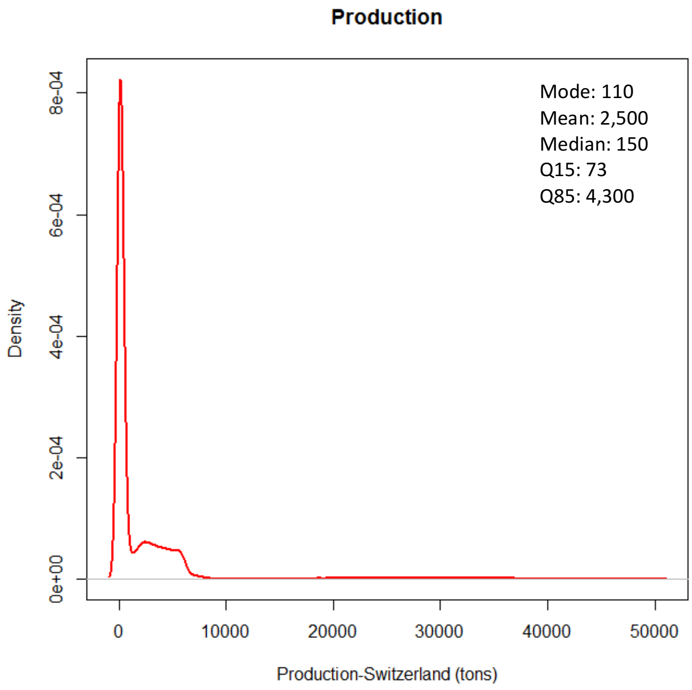


Figure S1.2. Probability distribution of nano-silica production (tons) in Switzerland in 2014. All values are rounded to two significant numbers

Ecotoxicological endpoints

A comprehensive literature review has been performed and twenty toxicity concentrations of nano-silica for aquatic species have been collected including half maximal effective concentration (EC_{50}), 20% effect concentration (EC_{20}), median lethal concentration (LC_{50}), 20% lethal concentration (LC_{20}), no observed effect concentration (NOEC), highest no-observed concentration (HNOEC) and the half maximal inhibitory concentration (IC_{50}). If a published paper investigated acute and sub chronic or chronic toxicity, the long-term toxicity concentration was chosen. NOEC values were preferred when other long-term toxicity data was also available. Those collected data were extrapolated to NOEC values by assessment factors. The table below presents sensitivity concentrations from species in different taxonomic groups and the applied assessment factors.

Table S1.7. The dataset of ecotoxicological endpoint concentrations of nano-silica

Item	*Toxic endpoint*	*Taxonomic group*	*Test organisms*	*Particle size (nm)*	*Concentration PEC [µg/l]*	*Exposure time [h]*	*Assessment Factor (AF) for the extrapolation from short to long term effect*	*Assessment Factor (AF) for the extrapolation from the observed effect into no effect concentrations*	*Species sensitivity [µg/l]*	*Source*
1	EC20	Algae	*Scenedesmus obliquus*	10 - 20	144'000	48	10	2	7'200	Wei et al., 2010 [18]
2	EC20	Algae	*Scenedesmus obliquus*	10 - 20	388'000	72	1	2	194'000	Wie et al., 2010 [18]
3	EC20	Algae	*Scenedesmus obliquus*	10 - 20	216'000	96	1	2	108'000	Wie et al., 2010 [18]
4	NOEC	Algae	*Pseudokirchneriella subspicata* , pH 6.8	22	10'000	48	10	1	1'000	Van Hoecke et al.,2011 [19]
5	NOEC	Algae	*Pseudokirchneriella subspicata*, pH 7.6	22	4'600	48	10	1	460	Van Hoecke et al., 2011 [19]
6	NOEC	Algae	*Pseudokirchneriella subspicata* , pH 8.6	22	10'000	48	10	1	1'000	Van Hoecke et al., 2011 [19]
7	NOEC	Algae	*Pseudokirchneriella subspicata*; NOM 1.2mg C/l	22	22'000	48	10	1	2'200	Van Hoecke et al., 2011 [19]
8	NOEC	Algae	*Pseudokirchneriella subspicata*; ,NOM 4.7mg C/l	22	100'000	48	10	1	10'000	Van Hoecke et al., 2011 [19]
9	NOEC	Algae	*Pseudokirchneriella subspicata*; ,NOM 9.0mg C/l	22	100'000	48	10	1	10'000	Van Hoecke et al., 2011 [19]
10	IC50	Algae	*Chlorella kessleri*	5	8'000'000	96	1	10	800'000	Fujiwara et al., 2008 [20]
11	IC50	Algae	*Chlorella kessleri*	26	71'000'000	96	1	10	7'100'000	Fujiwara et al., 2008 [20]
12	IC50	Algae	*Chlorella kessleri*	78	91'000'000	96	1	10	9'100'000	Fujiwara et al., 2008 [20]
13	LC50	Invertebrate	*Daphnia magna*		10'000'000	48	10	10	100'000	Nanosil (Asia Pacific) Sdn Bhd [21]
14	EC50	Invertebrate	*Daphnia magna*	10-20	148871	24	10	10	1'489	Shangyue Yang et al., 2015 [22]
15	LC50	Invertebrate	*Daphnia magna*	10-20	660'943	24	10	10	6'609	Shangyue Yang et al., 2015 [22]
16	HONEC	Unicellular	*Saccharomyces cerevisiae*		1'000'000	14	10	1	100'000	Garcia-Saucedo et al.,2011 [23]

17	LC50	Vertebrate	Goldfish	10-20	5'000'000	24	10	10	50'000	Nanosil (Asia Pacific) Sdn Bhd [21]
18	LC50	Vertebrate	*Poecilia reticulata*		10000000	48	10	10	100'000	Nanosil (Asia Pacific) Sdn Bhd [21]
19	HONEC	Vertebrate	*Danio rerio, zebrafish*	62	200'000	24	10	1	20'000	Karl et al.,2010 [24]
20	LC20	Vertebrate	Danio rerio, zebrafish	62	200000	96	10	2	10'000	Junchao Duan et a.,2013l [25]

Hazard characterization

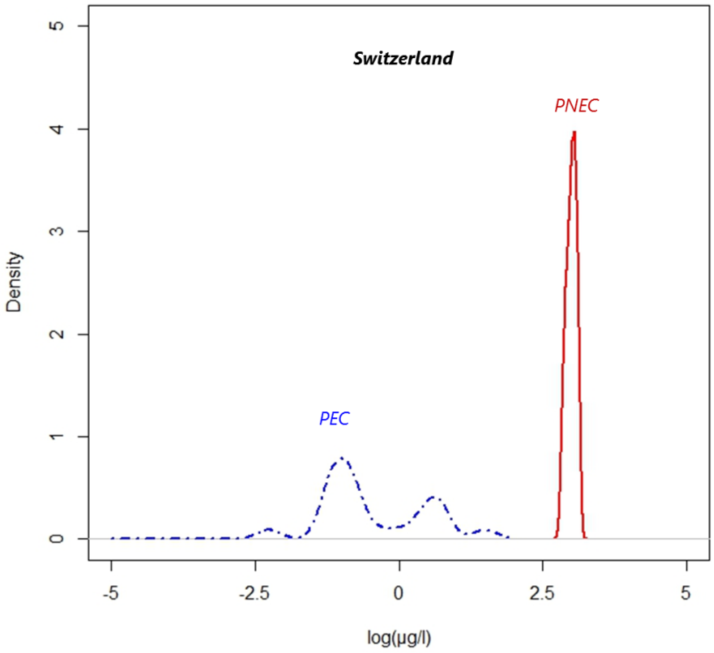


Figure S1.3. The Predicted No-Effect Concentration (PNEC) distribution compared with the probability density distribution of Predicted Environmental Concentrations (PEC) in Switzerland for nano-silica in surface water

Risk calculation

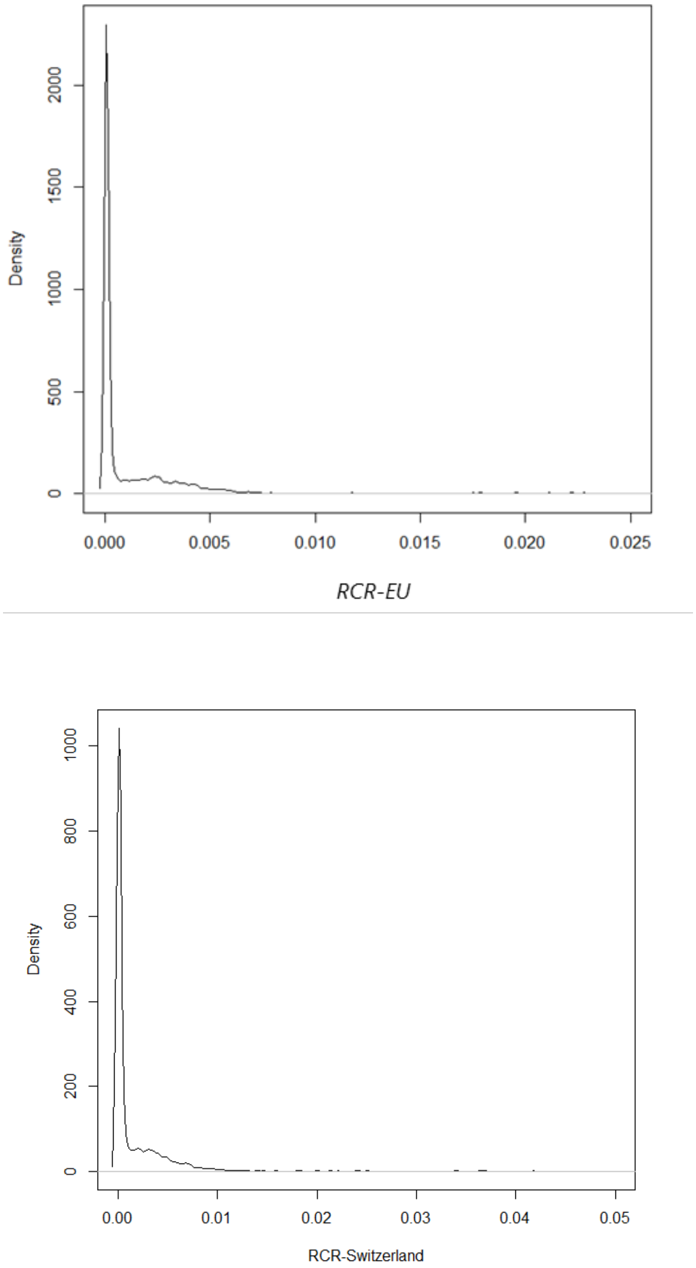


Figure S1.4. Risk quotient distributions for nano-silica in surface water in the EU (top) and Switzerland (bottom)

References:

1. Future Market, *The Global Market for Metal Oxide Nanoparticles 2014.* Future Market, 2014.

2. Piccinno, F., et al., *Industrial Production Quantities and Uses of Ten Engineered Nanomaterials for Europe and the World.* J. Nanopart. Res., 2012. **14**: p. 1109.

3. Michel, K., et al., *Risk assessment of amorphous silicon dioxide nanoparticles in a glass cleaner formulation.* Nanotoxicology, 2013. **7**: p. 974-88.

4. Schmid, K. and M. Riediker, *Use of nanoparticles in Swiss industry: A targeted survey.* Environ. Sci. Technol., 2008. **42**: p. 2253-2260.

5. ANSES, *Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)* 2013.

6. Willcocks, D., *NICNAS Information Sheet, Summary of call for information on the use of Nanomaterials; Australian Government.* 2007.

7. UNEP, *United Nations Environment Programme. Chapter 7: Emerging Challenges - Nanotechnology and the Environment, Geo Year Book.* 2007.

8. Liu, Y., et al., *Silica nanoparticles separation from water: Aggregation by cetyltrimethylammonium bromide (CTAB).* Chemosphere, 2013. **92**: p. 681-687.

9. OECD, *Initial Assessment Report for Synthetic amorphous silica and silicates* 2004.

10. Park, B., *Current and future applications of nanotechnology.* Issues in Environmental Science and Technology, 2007. **24**: p. 1-18.

11. Gottschalk, F. and B. Nowack, *Release of engineered nanomaterials to the environment.* J. Environ. Monitoring, 2011. **13**: p. 1145 – 1155.

12. Grass, R.N., et al., *Tracking Trace Amounts of Submicrometer Silica Particles in Wastewaters and Activated Sludge Using Silica-Encapsulated DNA Barcodes.* Environmental Science & Technology Letters, 2014. **1**: p. 484-489.

13. Jarvie, H.P., et al., *Fate of Silica Nanoparticles in Simulated Primary Wastewater Treatment.* Environmental Science & Technology, 2009. **43**: p. 8622-8628.

14. Den, W. and C.P. Huang, *Electrocoagulation of silica nanoparticles in wafer polishing wastewater by a multichannel flow reactor: A kinetic study.* J. Environ. Eng., 2006. **132**(12): p. 1651-1658.

15. Huang, C., W. Jiang, and C. Chen, *Nano silica removal from IC wastewater by pre-coagulation and microfiltration.* Wat. Sci. Technol., 2004. **50**: p. 133-138.

16. Pan, J.R., et al. *Treatment of wastewater containing nano-scale silica particles by dead-end microfiltration: evaluation of pretreatment methods*. in *Conference on Membranes in Drinking and Industrial Water Production*. 2004. L Aquila, ITALY.

17. Den, W. and C.P. Huang, *Electrocoagulation for removal of silica nano-particles from chemical-mechanical-planarization wastewater.* Colloids and Surfaces a-Physicochemical and Engineering Aspects, 2005. **254**: p. 81-89.

18. Wei, C.X., et al., *Effects of silica nanoparticles on growth and photosynthetic pigment contents of Scenedesmus obliquus.* Environmental Science 2010. **22**: p. 6.

19. Van Hoecke, K., et al., *Influence of alumina coating on characteristics and effects of SiO2 nanoparticles in algal growth inhibition assays at various pH and organic matter contents.* Environ Int, 2011. **37**: p. 1118-25.

20. Fujiwara, K., et al., *Size-dependent toxicity of silica nano-particles to Chlorella kessleri.* Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering, 2008. **43**: p. 1167-1173.

21. Nanosil, *Material Safety Data Sheet 4048. Nanosil® Nano Silica Dispersion MSDS.* p. 6.

22. Yang, S.Y., et al., *Ecotoxicological Effect of Nano-silicon Dioxide Particles on Daphnia Magna.* Integrated Ferroelectrics, 2014. **154**: p. 64-72.

23. Garcia-Saucedo, C., et al., *Low toxicity of HfO2, SiO2, Al2O3 and CeO2 nanoparticles to the yeast, Saccharomyces cerevisiae.* J Hazard Mater, 2011. **192**: p. 1572-9.

24. Fent, K., et al., *Assessment of uptake and toxicity of fluorescent silica nanoparticles in zebrafish (Danio rerio) early life stages.* Aquat Toxicol, 2010. **100**: p. 218-28.

25. Junchao Duan., et al., *Toxic effect of silica nanoparticles on zebrafish embryos and larvae.* PLOS ONE 2013. **8**: p. 9.

Supporting information for Annex 2

Production volume

Table S2.1 provides the data on production volume from six different sources and the DoB assigned to each source, as well as the GDP ppp values and the scaling factors from worldwide or country-specific value to Europe or Switzerland. Sources published before 2010 and after 2010 were assigned a DoB of 20 and 80 respectively. The French registration report was given a high DoB (80) because it is mandatory for enterprises to register the nano-FeO_X produced in or importedto France. Personal communication with experts from nanoparticle-producing industries was also regarded as a well-grounded source. Those papers and reports in which a detailed data collection method was described were considered as more reliable source and assigned with high DoB (80), otherwise 20 DoB was assigned. The decision tree for obtaining the DoB is displayed in Figure S2.2.

Table S2.1 Data on production volume and scaling factors

Production volume [tons/a]				**Reference**
80 DoB		20 DoB		
EU	CH*	EU	CH*	
2.7	0.07			[1]
		14600	365	[2]
5493-6991	137-174			[3]
16	0.4			[4]
		18-89	0.4-2.2	[5]
716-7163	18-179			[6]

Factors for scaling to different geographic regions:

According to GDP ppp (2014): (World Bank 2014)

World	**EU**	**Switzerland**	**Australia**	**France**
1.08463E+14	1.8423E+13	4.6061E+11	1.0313E+12	2.5720E+12

Scaling factors:

EU-CH	**World-CH**	**World-EU**	**EU-AUS**	**AUS-CH**	**FR-EU**	**FR-CH**
39.9970	235.4769	5.8874	17.8639	2.2390	7.1629	5.5839

*FeOx fraction in Fe and FeOx production is estimated to be 98%.

*CH here refers to Switzerland

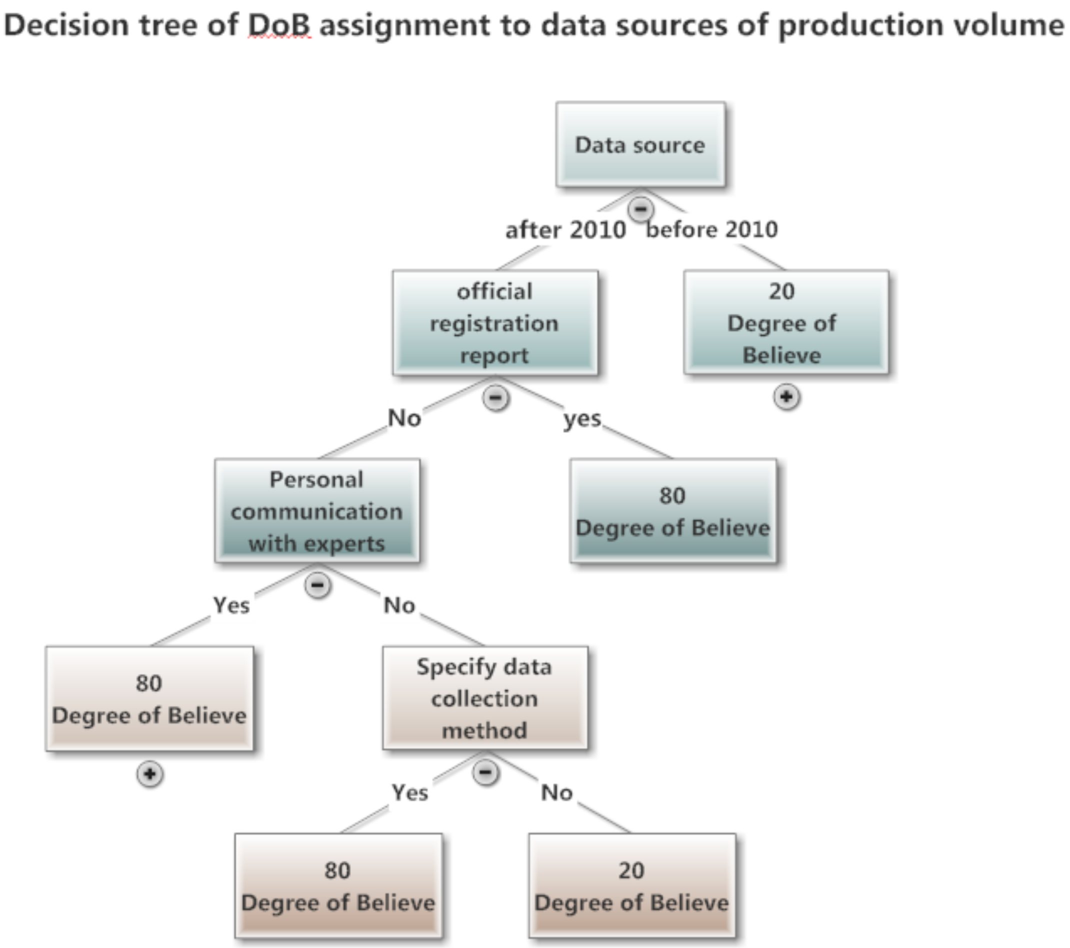


Figure S1. Decision tree of the DoB assigned to the data sources of the production volume

2. Product allocation

Because there is no comprehensive database available with complete information about products containing nano iron oxides, resources such as Future Market (2014), ANSES (2014), and CEH (Chemical Economics Handbook) were used because they have a relatively complete list of nano iron oxide applications. In order to avoid duplication of applications, a classification was needed. According to a RIVM report [7], the proposed list of product categories covers applications from appliances, electronics and motor vehicles to health products, sporting goods, textiles and toys. Moreover, the French registration report also states the detailed applications of nano-FeO_X based on the registration information provided by companies [6]. In our study, the classification was based on the information from the aforementioned reports. Four methods were used in the study to conduct the product allocation.

2.1 Method 1 (Future Market 2012)

Based on the market study by Future Market, Keller et al. have built the global material flows for nano-FeO_X and nano zero valent (NZVI) iron in 2010 assuming maximum production and emissions rates [1]. The application of NZVI and nano iron oxides comprised electronics & optics, catalysts, coatings, paints & pigments, medical, aerospace, cosmetics, energy and environment. In our study, since we only consider nano-FeO_X, an exclusion of NZVI from the reported flows was required. A search of NZVI and nano-FeO_X applications from two databases (Google scholar and Web of Science) was conducted to find out where NZVI was used. The result indicated that the use of NZVI was mainly focused on the environmental field. However, the information regarding applications of nano-FeO_X, covering electronics, medicals, coatings, and catalyst were much more diversified than NZVI. Hence, we assumed that in the Future Markets report the "Energy & Environment" category consists only of NZVI. Thus, we have removed the share of this category from our evaluation which is specific for nano-FeO_X

2.2 Method 2.1 (US patent database) and Method 2.2 (EU patent database)

The US patent database by the United States Patent and Trademark Office maintains a permanent, interdisciplinary historical record of all U.S patent applications [8]. The EU patent database by the European Patent Register contains all the publicly available information on European patent applications as they pass through the grant procedure [9]. Through a keyword search “nano iron oxide” for each database, the number of products containing nano iron oxides and their applications can be found. After counting the number of products for each application category, the share of each application category can be obtained for each database.

2.3 Method 3 (Global Market)

The "Global market method" defines the percentage of nano-FeO_X in each product categories based on the overall value of each defined commercial product. First of all, the global market value for each application category was collected. Then the average market price for the corresponding product was gathered. The third step was to find out the average weight of the product and its nano iron oxides fraction through a literature search. The total quantity of a certain application was obtained by dividing the global market value for each application by the average price of the product. Multiplying the quantity of each product category by the nano iron oxide fraction of that, we can get the amount of total nano-FeO_X in each category. For categories, of which the quantity is hard to characterize, the price per ton is applied instead of the average price of the product.

2.4 Method 4 (Production volume collection)

The production volume method takes advantages of data on the amounts of nano-FeO_X used in a certain product category from different sources [10, 11]. When data is a direct representation of the production amount, it can be used without further treatment. When data indicates the production amount, however, without clear explanation of its application or just on behalf of large particles, further processing is required. For example, the Danish EPA (Danish EPA 2015) has information about the production volume of iron oxide pigments as well as the composition of their respective categories. At the same time, CEH report (CEH 2011) implies that nano-sized iron oxides account for 4% of the total amount of about 20 thousand metric tons in 2012. After multiplying the production volume for each pigment category by the share, nano-sized iron oxides production volume for each category is obtained.

In this study, a DoB was given to each method according to its reliability. Method 1 and method 3 were based on the market value of each application, therefore they were considered as methods of higher credibility (30). We gave low DoB to Method 2 (10), because the patent applications might be not on the market yet. 20 of DoB was chosen for Method 4, given the fact that it used data, which might not represents directly the amount of nano-FeO_X used in a certain product category.

2.5 Results for product distribution

Table S2 shows the summary of the data collection for the product allocation. Based on the reliability for each method, a different DoB was given. M1 (Future Market 2012 method), M3 (Market size method) and M4 (Production volume collection method) were based on the production amount and market value for each category. However, M2.1 (US patent database method) and M2.2 (EU patent database method) were based on the patent publication numbers of each application. M1, M3 and M4 were considered to be more proper for material flow modeling. A higher DoB was therefore allocated accordingly. However, it is also important to consider M2.1 and M2.2 as an indicator of the possible use of nano-FeO_X, even it is not guaranteed that those new patents are applied in the market. A lower DoB was correspondingly assigned to M2.

Table S2. The allocation of nano iron oxides to different product categories according to M1-M4 methods

Product Category	DoB					Weighted mean
	30	10	10	30	20	
	M1	M2.1	M2.2	M3	M4	
Catalyst	13.0%	-	-	12.0%	0.9%	7.7%
Ceramics	-	-	-	-	0.7%	0.1%
Paints	34.0%	21.0%	-	50.0%	51.3%	37.6%
Construction	-	63.0%	-	-	32.9%	12.9%
Cosmetics	13.0%	-	-	16.0%	0.6%	8.8%
Electronics	25.0%	11.0%	-	22.0%	3.9%	16.0%
Food	-	-	-	-	1.4%	0.3%
Paper	-	-	-	-	1.9%	0.4%
Plastics	-	5.0%	-	-	4.5%	1.4%
Sands	-	-	-	-	1.9%	0.4%
Aerospace	2.0%	-	-	-	-	0.6%
Battery	-	-	50.0%	-	-	5.0%
Medical	13.0%	-	25.0%	-	-	6.4%
Water	-	-	25.0%	-	-	2.5%

3. *Transfer coefficients*

Table S3. Transfer Coefficients from products to compartments

Category	Wastewater		WIP		Landfill		Air		Soil		Surface water		Recycling		Export		Reference
	EU	CH	EU	CH	EU	CH	EU	CH	EU	CH	EU	CH	EU	CH	EU	CH	
Production	0.75	0.75					0.25	0.25									[12]
Manufacture	0.33	0.33	0.11	0.33	0.23		0.335	0.335									[12]
Medical*	0.95	0.95	0.05	0.05													
Cosmetics	0.85	0.85	0.02	0.05	0.03						0.1	0.1					[12]
Electronics			0.05	0.05	0.1								0.65	0.75	0.2	0.2	[12]
Catalyst			0.05	0.05									0.95	0.95			[13]
Paints	0.02	0.02			0.50	0.50	0.01	0.01			0.01	0.01	0.46	0.46			[12]
Aerospace	0.05	0.05	0.02	0.05	0.03								0.9	0.9			[12]
Construction	0.01	0.01			0.29	0.19							0.7	0.8			[14]
Food	0.9	0.9	0.03	0.1	0.07												[12]
Paper			0.05	0.1	0.05								0.8	0.8	0.1	0.1	[12]
Plastics				0.38	0.38								0.62	0.62			[15]
Ceramics	0.01	0.01	0.05	0.2	0.15								0.79	0.79			[12]
Sands					1	1											[16]
Battery			0.04	0.13	0.09								0.33	0.33	0.54	0.54	[12]
Water *	0.5	0.5							0.4	0.4	0.1	0.1					

* Expert judgement

Table S4 Nano iron oxide removal efficiency in WWTP and corresponding Degree of Belief (DoB)

Removal efficiency [%]	DoB	Comments	Reference
69%-92%	20	Standard Jar test (experimental): Total iron removal efficiency when different combinations of ferric chloride and polymer ratio were applied during coagulation and flocculation process.	[17]
95%-98%	30	Model WWTP: TiO_2 removal efficiency adapted from Sun, et al. 2014	[18]
96.1%-99.4%	30	Full scale WWTP for titanium removal	[19]
96%-98%	50	Simulated WWTP TiO_2 nanomaterials removal	[20]
89.5%	50	Real wastewater treatment plant: (TiO_2) A removal of 89.5% of the Ti<0.45μm.	[21]
96%-97%	50	Laboratory sewage treatment plant (LSTP): nanoscale TiO_2 (<250nm) Removal efficiency has been assessed.	[22]

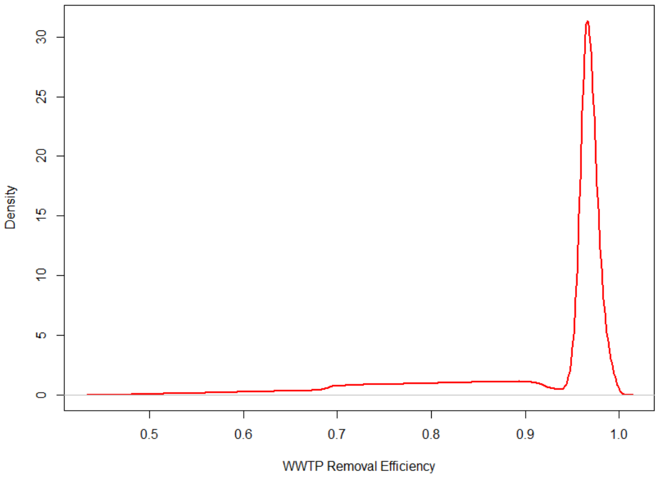


Figure S2. Probability distribution of WWTP removal efficiency for nano iron oxides

4. Flows

Table S5. Mass flows of nano-FeO_X to technical and environmental compartments for the EU and Swiss systems (Tons/year)

	EU				CH			
	Q15	**Median**	**Mean**	**Q85**	**Q15**	**Median**	**Mean**	**Q85**
From PMC to wastewater	0.39	120	1000	1300	8.8E-03	4.7	25	32
From PMC to landfill	0.69	220	1700	2200	1.6E-02	8.5	46	57
From PMC to WIP	0.45	140	1100	1400	1.1E-02	5.4	29	36
From PMC to recycling	1.2	380	3100	3800	2.6E-02	14	75	93
From PMC to export	0.16	51	410	520	3.6E-03	2.1	11	13
From PMC to air	0.018	5.8	46	59	4.1E-04	0.22	1.2	1.6
From PMC to soil	0.028	8.1	69	91	6.3E-04	0.34	1.8	2.2
From PMC to surface water	0.041	13	103	130	9.2E-04	0.51	2.6	3.3
From Wastewater to surface water	0.081	24	200	260	2.3E-04	0.13	1	0.84
From Waste water to WWTP	0.32	96	800	1020	8.6E-03	4.6	24	31
From Bottom ash to recycling	0.24	63	590	770	0	0	0	0
From Bottom ash to landfill	0.33	90	820	1050	1.6E-02	5.4	45	58
From Filter to filter ash (WIP removal efficiency)	0.1402369	37	330	440	4.0E-03	1.3	11	14
From Filter to air	2.1E-04	4.3E-02	5.1E-01	6.8E-01	0	0	0	0
From Filter to acid washing 2	2.2E-04	4.5E-02	5.4E-01	7.1E-01	1.3E-05	3.1E-03	3.3E-02	4.5E-02
From Filter ash to export	-	-	-	-	8.8E-04	0.28	2.4	3.2
From Filter ash to landfill	0.14	37	330	440	1.6E-03	0.5	4.3	5.6
From Filter ash to acid washing 1	-	-	-	-	1.6E-03	0.5	4.3	5.6
From Acid washing 2 to air	1.6E-07	3.6E-05	3.9E-04	5.2E-04	1.2E-08	3.0E-06	3.3E-05	4.5E-05
From Acid washing 2 to landfill	1.7E-04	3.8E-02	3.9E-01	5.2E-01	1.3E-05	3.1E-03	3.3E-02	4.5E-02
Overflow	0.011	3.2	25	33	2.8E-04	0.15	0.79	0.99
Effluent	1.4E-03	3.5	67	70	3.9E-04	0.11	2.0	2.1
From Sludge to landfill	5.6E-02	16	140	180	-	-	-	-
From Sludge to WIP	7.0E-02	21	180	220	5.9E-03	3.2	17	21
From Sludge to cement plant	-	-	-	-	0.0017	0.89	4.7	5.9
From Sludge to soil	0.15	45	390	500	-	-	-	-
From Air to surface water	5.6E-04	0.18	1.4	1.8	1.7E-05	9.2E-03	4.9E-02	6.1E-02
From Air to soil	0.02	5.7	45	57	3.9E-04	0.21	1.1	1.4
From Soil to surface water	1.1E-03	0.32	2.7	3.5	5.6E-06	3.0E-03	1.6E-02	2.0E-02
Recycling (considered)	1.10	305.0	2600	3300	2.2E-02	7.7	64	82
From Recycling to WIP	0.18	49	440	560	0.0039	1.3	11	14
From Recycling to landfill	0.68	200	1700	2200	0.015	5.1	42	54
From Recycling to PMC	0.1	24	210	260	0.0018	0.6	5.3	6.6
From Recycling to cement	0.041	12	103	130	0.001	0.3	2.6	3.3
From Recycling to export	0.061	15	150	200	0.0013	0.41	3.6	4.8

Note: PMC refer to 'Production, Manufacturing and Consumption' compartment described in the main text

Table S6. Ecotoxicological endpoint concentrations for nano iron oxides

	Environmental compartment	**Taxonomic group**	***Test Organism***	**Particle Size (nm)**	**Test material**	**Test condition**	**Exposure time (h)**	**Test endpoint**	**Ecotoxic endpoint**	**Concentration (µg/l)**	**Assessment Factor (AF) for the extrapolation from short to long term effect**	**Assessment Factor (AF) for the extrapolation from the observed effect into no effect concentrations**	**Species sensitivity [µg/l]**	**Reference**
1	Water	Unicellular	*Escherichia coli (Migula)*	30-40	Fe2O3		2	Mortality	LC50	638300	10	10	6383	[23]
2	Water	Unicellular	*Escherichia coli*	8.0-21.0	Fe3O4 suspension	25 ºC	24	Inhibition	LOEC	100000	10	2	5000	[24]
3	Water	Unicellular	*Photobacterium phosphoreum*	7.57 ± 5.6	Fe3O4	pH 7	0.083-0.25	Bioluminescence reduction	EC50	> 52000	10	10	>520	[25]
4	Water	Unicellular	*Vibrio fischeri*	6	Fe3O4 solution		0.083-0.25	Light output reduction	EC50	240000	10	10	2400	[26]
5	Water	Unicellular	*Vibrio fischeri*	8.0-21.0	Fe3O4 suspension	20 ºC	0.5	Luminescence inhibition	EC50	>100000	10	10	1000	[24]
6	Water	Vertebrate	*Oryzias latipes*	27.2 ± 1.0	Fe3O4		168	Mortality	HNOEC	87200	10	1	8720	[27]
7	Water	Invertebrate	*Ceriodaphnia dubia*	21-890	Fe2O3	pH 7.8	24	Mortality	NOEC	200000	10	1	20000	[28]
8	Water	Invertebrate	*Ceriodaphnia dubia*	21-890	Fe2O3	pH 7.8	48	Mortality	NOEC	200000	10	1	20000	[28]
9	Water	Vertebrate	*Danio rerio (embryo)*	30	α-Fe2O3 NPs	26 ± 1°C	168	Survival	NOEC	<50000	10	1	5000	[29]
10	Water	Vertebrate	*Danio rerio (egg)*	<50	Fe2O3	pH 8.3–8.6.	48	Mortality	LC50	>1600000	10	10	16000	[30]
11	Water	Vertebrate	*Danio rerio (adult)*	<50	Fe2O3	pH 8.3–8.6.	48	Mortality	LC50	>1600000	10	10	16000	[30]
12	Water	Vertebrate	*Danio rerio (egg)*	<50	Fe2O3	pH 8.3–8.6.	96	Mortality	LC50	>1600000	10	10	16000	[30]
13	Water	Vertebrate	*Danio rerio (adult)*	<50	Fe2O3	pH 8.3–8.6.	96	Mortality	LC50	>1600000	10	10	16000	[30]
14	Water	Invertebrate	*Daphnia magna*	6	Fe3O4 solution		48	Mortality	LC50	2300	10	10	23	[26]
15	Water	Invertebrate	*Daphnia magna*	5.7 ± 0.5	PVP-coated magnetite (Fe3O4)	80 ºC	96	Immobilization	LOEC	100000	10	2	5000	[31]
16	Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Ascorbate-Fe3O4 coating	pH 7	96	Immobilization	EC50	50700	10	10	507	[32]
17	Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Citrate-Fe3O4 coating	pH 7	96	Immobilization	EC50	68400	10	10	684	[32]
18	Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Dextran-Fe3O4 coating	pH 7	96	Immobilization	EC50	27900	10	10	279	[32]

19	Water	Invertebrate	*Daphnia magna*	6.1 ± 0.6	Polymer-Fe3O4 coating	pH 7	96	Immobilization	EC50	> 100000	10	10	1000	[32]
20	Water	Invertebrate	*Daphnia magna*	40-130	Fe2O3	20±2 °C	48	lethal effect	HONEC	20000	10	1	2000	[33]
21	Water	Invertebrate	*Brachionus plicatilis*	40-130	Fe2O3	20±2 °C	48	growth rate	HONEC	20000	10	1	2000	[33]
22	Water	Unicellular	*Pseudokirchneriella subcapita*	40-130	Fe2O3	20±2 °C	72	growth rate	HONEC	20000	1	1	20000	[33]
23	Water	Unicellular	*Pseudokirchneriella subcapita*	8.0-21.0	Fe3O4 suspension	24 ± 1 °C	72	Algal growth inhibition	EC50	1930	1	10	193	[24]
24	Water	Invertebrate	*Lemna minor*	40-130	Fe2O3	20±2 °C	7 days	growth rate	HONEC	20000	10	1	2000	[33]
25	Water	Unicellular	*Tetrahymena thermophila*	8.0-21.0	Fe3O4 suspension	25ºC	24	Inhibition	EC50	26030	10	10	260.3	[24]
26	Water	Unicellular	*Staphylococcus aureus*	8.0-21.0	Fe3O4 suspension	25ºC	24	Inhibition	LOEC	100000	10	2	5000	[24]

Fehler! Keine gültige Verknüpfung.

Fehler! Keine gültige Verknüpfung.

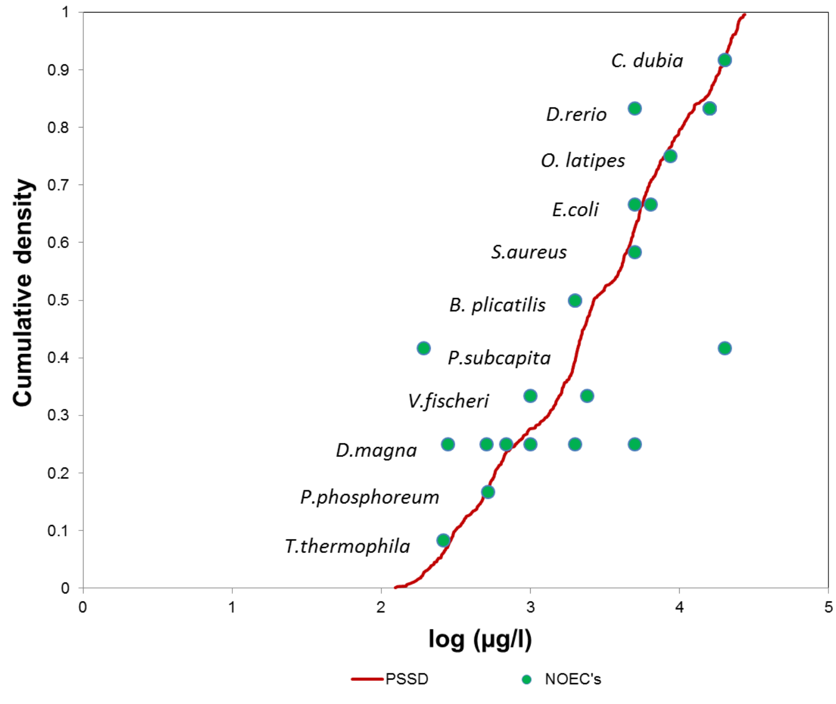


Figure S3. PSSD from sensitivity analysis (excluding the lowest NOEC of D. magna)

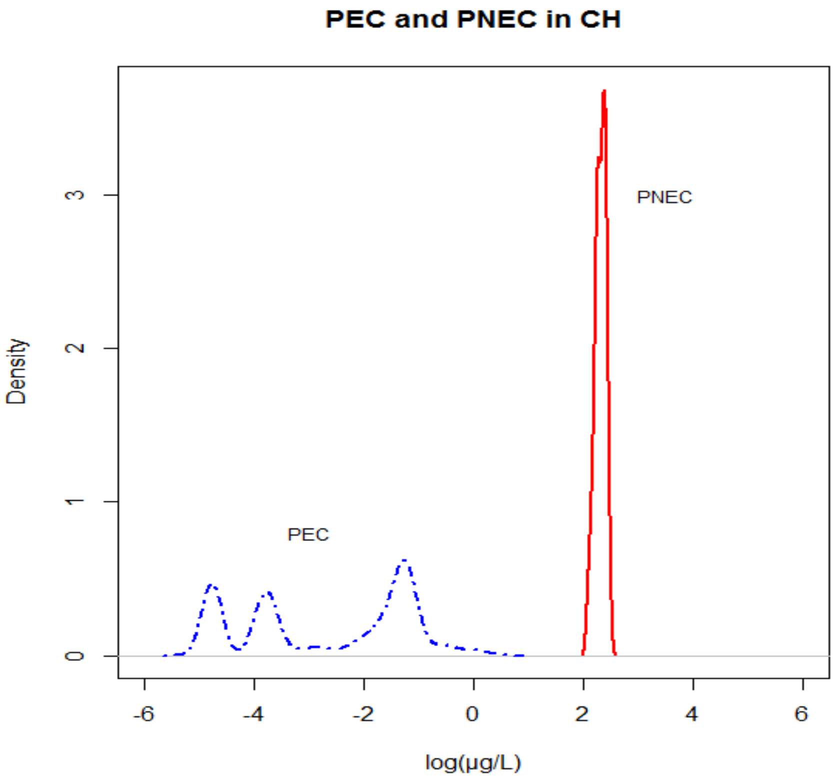


Figure S4. **Predicted environmental**

concentration distribution (in blue) and predicted no-effect concentration distribution (in red) for the Swiss system.

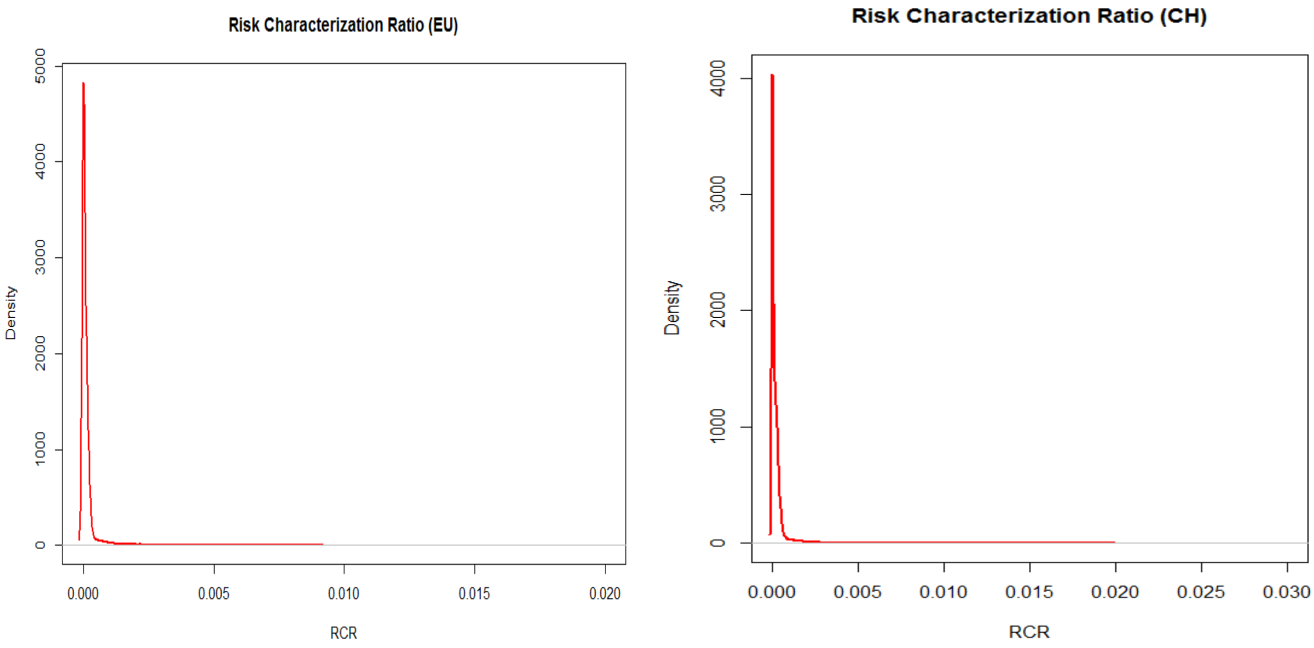


Figure S5. Risk characterization ratio distribution for the EU (left) and Switzerland (right)

References

1. Future Market, *The Global Market for Metal Oxide Nanoparticles 2014.* Future Market, 2014.
2. Schmid, K. and M. Riediker, *Use of nanoparticles in Swiss industry: A targeted survey.* Environ. Sci. Technol., 2008. 42: p. 2253-2260.
3. Future Markets, *The global market for nanomaterials 2002–2006: production volumes, revenues and end use markets. Future Markets Inc., http://www.futuremarketsinc. com/index.php?option=com_content&view=article&id= 176&Itemid=73.* 2012.
4. *COMMISSION STAFF WORKING PAPER : Types and uses of nanomaterials, including safety aspect.* 2012.
5. Willcocks, D., *NICNAS Information Sheet, Summary of call for information on the use of Nanomaterials; Australian Government.* 2007.
6. ANSES, *Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)* 2014.
7. Pronk, M.E.J., et al., *Nanomaterials under REACH. Nanosilver as a case study. RIVM report 601780003, 76 pp. .* 2009.
8. USP&TO, *United States Patent and Trademark Office. Patent Full-Text Databases Available from: http://patft.uspto.gov/netahtml/PTO/index.html. Retrieved 14.03.2015.* 2015.
9. EPR, *European Patent Office. European Patent Register Available from: http://www.epo.org/index.html. Retrieved 14.03.2015.* 2015.
10. SRI, *The Chemical Economics Handbook: Inorganic chemical pigments*. p. 40.
11. Mikkel Aaman Sørensen, et al., *Survey of products with nanosized pigments*. 2015.
12. Sun, T.Y., et al., *Comprehensive Modeling of Environmental Emissions of Engineered Nanomaterials.* Environ. Pollut., 2014. 185: p. 69-76.
13. Wang, Y., et al., *Probabilistic modeling of the flows and environmental risks of nano-silica.* Sci Total Environ, 2016. 545-546: p. 67-76.
14. Hincapie, I., et al., *Use of engineered nanomaterials in the construction industry with specific emphasis on paints and their flows in construction and demolition waste in Switzerland.* Waste Management, 2015. 43: p. 398-406.
15. PlasticsEurope, *Plastics – the Facts 2015 :An analysis of European plastics production, demand and waste data.* 2015: p. 30.
16. Administration, U.S.D.o.T.-F.H., *User Guidelines for Waste and Byproduct Materials in Pavement Construction.* 2016.
17. Amuda, O.S. and I.A. Amoo, *Coagulation/flocculation process and sludge conditioning in beverage industrial wastewater treatment.* J Hazard Mater, 2007. 141(3): p. 778-83.
18. Limbach, L.K., et al., *Removal of oxide nanoparticles in a model wastewater treatment plant: Influence of agglomeration and surfactants on clearing efficiency.* Environ. Sci. Technol., 2008. 42(15): p. 5828-5833.
19. Westerhoff, P., et al., *Occurrence and removal of titanium at full scale wastewater treatment plants: implications for TiO(2) nanomaterials.* Journal of Environmental Monitoring, 2011. 13(5): p. 1195-1203.
20. Wang, Y., P. Westerhoff, and K.D. Hristovski, *Fate and biological effects of silver, titanium dioxide, and C60 (fullerene) nanomaterials during simulated wastewater treatment processes.* J Hazard Mater, 2012. 201-202: p. 16-22.
21. Johnson, A.C., et al., *An assessment of the fate, behaviour and environmental risk associated with sunscreen TiO2 nanoparticles in UK field scenarios.* Science of the Total Environment, 2011. 409(13): p. 2503-2510.
22. Gartiser, S., et al., *Behavior of nanoscale titanium dioxide in laboratory wastewater treatment plants according to OECD 303 A.* Chemosphere, 2014. 104: p. 197-204.
23. Hu, X., et al., *In vitro evaluation of cytotoxicity of engineered metal oxide nanoparticles.* Sci Total Environ, 2009. 407(8): p. 3070-2.
24. Aruoja, V., et al., *Toxicity of 12 metal-based nanoparticles to algae, bacteria and protozoa.* Environmental Science-Nano, 2015. 2(6): p. 630-644.
25. Barrena, R., et al., *Evaluation of the ecotoxicity of model nanoparticles.* Chemosphere, 2009. 75(7): p. 850-7.

26. Garcia, A., et al., *Acute toxicity of cerium oxide, titanium oxide and iron oxide nanoparticles using standardized tests.* Desalination, 2011. 269(1-3): p. 136-141.

27. Chen, P.J., S.W. Tan, and W.L. Wu, *Stabilization or Oxidation of Nanoscale Zerovalent Iron at Environmentally Relevant Exposure Changes Bioavailability and Toxicity in Medaka Fish.* Environmental Science & Technology, 2012. 46(15): p. 8431-8439.

28. Hu, J., et al., *Quantifying the effect of nanoparticles on As(V) ecotoxicity exemplified by nano-Fe2O3 (magnetic) and nano-Al2O3.* Environmental Toxicology and Chemistry, 2012. 31(12): p. 2870-2876.

29. Zhu, X., S. Tian, and Z. Cai, *Toxicity assessment of iron oxide nanoparticles in zebrafish (Danio rerio) early life stages.* PLoS One, 2012. 7: p. e46286.

30. Kovriznych, J.A., et al., *Long-term (30 days) toxicity of NiO nanoparticles for adult zebrafish Danio rerio.* Interdiscip Toxicol, 2014. 7(1): p. 23-6.

31. Filser, J., et al., *Intrinsically green iron oxide nanoparticles? From synthesis via (eco-)toxicology to scenario modelling.* Nanoscale, 2013. 5(3): p. 1034-46.

32. Baumann, J., et al., *The coating makes the difference: acute effects of iron oxide nanoparticles on Daphnia magna.* Sci Total Environ, 2014. 484: p. 176-84.

33. Nogueira, V., et al., *Assessing the ecotoxicity of metal nano-oxides with potential for wastewater treatment.* Environ Sci Pollut Res Int, 2015. 22(17): p. 13212-24.

Supporting information for Annex 3

1. Geographical systems

In this study, seven systems were considered, which are the EU (EU 27), central Europe, east Europe, south Europe, north Europe, South east Europe, and Switzerland. The corresponding countries of each region are listed in table below (EEA. 2013).

Table S3.1. Geographical systems and the corresponding countries

System	Countries
EU	EU- 27
Central Europe	Austria, Belgium, United Kindom, Denmark,Germany, Luxemberg, Netherlands, Ireland, Switzerland
Northern Europe	Finland, Iceland, Norway, Sweden
Eastern Europe	Czech Republic, Estonia, Hungary, Poland, Slovak Republic, Slovenia, Latvia, Lithuania
Southern Europe	Cyprus, Portugal, Greece, France, Spain, Italy, Malta
Southeastern Europe	Turkey, Bulgaria, Romania
CH	Switzerland

2. Production data

Table S2 gives the collected data regarding the production volume of nano-SiO_2, nano iron oxides (nano FeOX), nano-CeO_2, nano Al_2O_3, and quantum dots (QDs). Each data source was assigned with a Degree of Belief (DoB), and the criteria of giving a DoB is given in Figure S1.

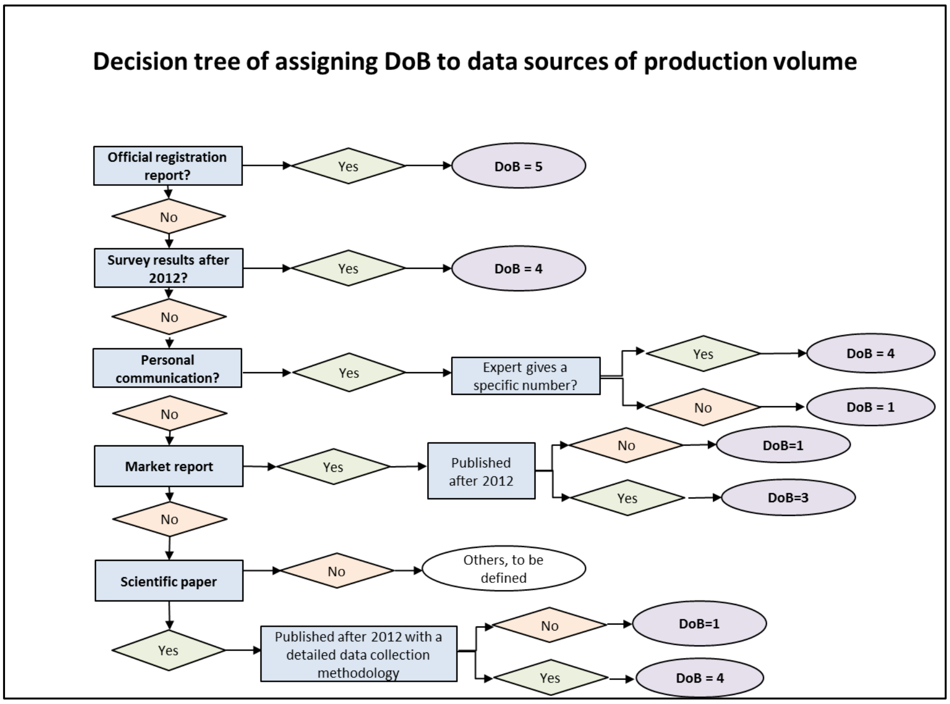


Figure S3.1. Decision tree of assigning degree of belief (DoB) to data sources of production volume

Table S3.2. The production volume of five ENMs in seven systems and the scaling factors

Production volume for nano-SiO_2 (Tonnes/year)

Reference year	EU	Switzerland	Central Europe	Northern Europe	Southern Europe	Eastern Europe	Southeastern Europe	DoB	Reference
2009	423757	11015	205397	22884	157080	43314	45908	1	(Future Markets 2014)
2010	399771	10392	193771	21588	148189	40863	43310	1	(Anonymous 2012)
2012	5500	75	2666	297	2039	562	596	4	(Piccinno et al. 2012)
2012	25319	658	12272	1367	9385	2588	2743	3	(Future Markets 2012)
2013	1116799	29030	541318	60310	413981	114154	120990	5	(ANSES 2013)
2014	>720186	>18720	>349078	>38892	>266962	>73614	>78022	5	(ANSES 2014)
2014	49305-373119	1282-9699	23898-180853	2663-20149	18277-138310	5040-38138	5342-40422	3	(Future Markets 2014)
2015	267-2665	7.0-69	129-1292	14-144	99-988	27-272	29-289	4	(Wang 2016)

Production volume for nano-FeOX (Tonnes/year)

Reference year	EU	Switzerland	Central Europe	Northern Europe	Southern Europe	Eastern Europe	Southeastern Europe	DoB	Reference
2007	18-89	0.47-2.0	9.0-43	1.0-5.0	7.0-33	2.0-9.0	2.0-10.0	1	(Willcocks 2007)
2008	14041	365	6806	758	5205	1435	1521	1	(Schmid &Riediker 2008)
2010	34	1	16	2	13	3	4	1	(Commission 2012)
2012	550	14	267	30	204	56	60	4	(Piccinno et al. 2012)
2012	14176	368	6871	766	5255	1449	1536	3	(Future Markets 2012)
2014	2.0-16.0	0.05-0.42	1.0-8.0	0.11-1	1.0-6.0	0.2-2	0.22-2	3	(Future Market 2014)
2014	720-7202	19-187	349-3491	39-389	267-2670	74-736	78-780	5	(ANSES 2015)
2015	344-3444	8.9-89	167-1670	18.6-186	127-1270	35-350	37-370	4	(Wang 2016)

Production volume for nano-CeO_2 (Tonnes/year)

Reference year	EU	Switzerland	Central Europe	Northern Europe	Southern Europe	Eastern Europe	Southeastern Europe	DoB	Reference
2010	2'507	65	1'215	135	929	256	272	1	(Commission 2012)
2012	2507	65	1215	135	929	256	272	3	(Future Markets 2012)
2012	55	1	27	3	20	6	6	4	(Piccinno et al. 2012)
2013	771	20	374	42	286	79	84	5	(ANSES 2013)
2013	1880	49	911	102	697	192	204	3	(Future Markets 2014)
2014	208-354	5.0-9.0	101-172	11.0-19.0	77-131	21-36	23-38	3	(Future Markets 2014)
2014	72-720	2.0-19.0	35-349	4.0-39	27-267	7.0-74	8.0-78	5	(ANSES 2015)
2015	2.78-27.8	0.072-0.72	1.3-13	0.15-1.5	1.0-10.3	0.13-1.3	0.3-3	4	(Wang 2016)

Production volume for nano-Al_2O_3 (Tonnes/year)

Reference year	EU	Switzerland	Central Europe	Northern Europe	Southern Europe	Eastern Europe	Southeastern Europe	DoB	Reference
2010	56'509	1'469	27'390	3'052	20'947	5'776	6'122	1	(Commission 2012)
2010	2119	55	1027	114	786	217	230	1	(Future Markets 2014)
2011	1695	44	822	92	628	173	184	1	(Future Markets 2014)
2012	550	14	267	30	204	56	60	4	(Piccinno et al. 2012)
2012	9889	257	4793	534	3666	1011	1071	3	(Future Markets 2012)
2013	15798	411	7657	853	5856	1615	1711	5	(ANSES 2013)
2014	1335-2857	35-74	647-1385	72-154	495-1059	136-292	145-310	3	(Future Markets 2014)
2014	7202-72019	187-1872	3491-34908	389-3889	2670-26696	736-7361	780-7802	5	(ANSES 2015)
2015	283	7	137	15	105	29	31	4	Expert judgement

Production volume for Quantum Dots (Tonnes/year)

Reference year	EU	Switzerland	Central Europe	Northern Europe	Southern Europe	Eastern Europe	Southeastern Europe	DoB	Reference
2012	104	3	50	6	38	11	11	4	(Piccinno et al. 2012)
2014	255	7	124	14	95	26	28	3	(Shreyas Naidu &Gotsurve 2014)
2015	170-1700	16561	82-820	9.2-92	63-630	17.4-174	18.4-184	4	(Wang 2016)

GDP PPP in 2014 ($), DoB (1)									
World	EU	CH	France	CE	NE	SE	EE	SEE	
1.09E+14	1.88E+13	4.88E+11	2.60E+12	9.09E+12	1.01E+12	6.95E+12	1.92E+12	2.03E+12	
Scaling factor									
World-CH	EU-CH	France-CH	World-EU	EU-CE	EU-NE	EU-SE	EU-EE	EU-SEE	EU-France
222.93	38.47	5.34	5.79	2.06	18.52	2.70	9.78	9.23	7.20

Regional market share, DoB (4)			
North America (NA)	Far East (FE)	Europe (EU)	Africa, Middle East& South America (Other)
0.15	0.52	0.31	0.02
Scaling factor			
World- NA	World- EU	World- FE	World- Other
6.66	3.22	1.92	50

EU: EU 27, CH: Switzerland, CE: Central Europe, EE: Eastern Europe, SE: Southern Europe, NE: Northern Europe, SEE: South-eastern Europe

3. *Product allocation*

Eight different methods, as displayed in Table S3, were used to calculate the product allocation for five considered ENMs. However, each method is not available for every ENM, which indicated as NA in the following table. In this paper, two methods, 'Market method' and 'Production volume' method, are used in additional to the approach we used in our previous work (Wang et al. 2016a, Wang et al. 2016b).

Table S3.3. The allocation of five ENMs to defined product categories

(1) nano-SiO2									
Category	**WWI** (WWI; 2012)	**B2B** (EC21 2015)	**BUND** (BUND 2011)	**Nanodb** (Nanodatabase 2013)	**Patent**(EPR 2015, USP&TO 2015)	**Market method**	**Future market**	**Production volume**	**Weigted Mean (%)**
DoB	**3**	**3**	**1**	**3**	**1**	**5**	**-**	**-**	
Adhesive	0.00	0.00	0.00	0.00	0.26	0.00	NA	NA	0.02
Automotive coating	5.39	5.52	1.14	3.63	2.37	5.70	NA	NA	4.73
Car tyre	25.5	58.7	0.00	0.00	0.00	7.08	NA	NA	17.99
Catalyst	0.00	0.00	0.00	0.00	2.80	0.00	NA	NA	0.18
Ceramics	3.86	0.00	0.00	0.00	0.42	0.00	NA	NA	0.75
Cleaning agent	18.04	0.00	0.00	0.00	0.00	2.16	NA	NA	4.06
Composite material	0.00	0.00	0.00	0.00	0.90	0.00	NA	NA	0.06
Construction material	7.93	0.00	2.52	0.00	6.54	0.00	NA	NA	2.05
Cosmetics & Personal care	1.13	4.35	2.70	3.43	0.09	0.81	NA	NA	2.10
Dental	0.10	0.08	0.00	0.00	0.13	0.00	NA	NA	0.04
Electrical cable	0.00	0.00	0.00	0.00	0.40	0.00	NA	NA	0.03
Electronic	0.07	0.00	0.01	0.00	0.10	0.00	NA	NA	0.02
Food & Supplement	2.89	0.00	12.24	1.17	0.00	0.00	NA	NA	1.53
Glass	1.29	0.00	0.00	1.04	0.00	0.00	NA	NA	0.44
Ink	0.00	0.00	0.00	0.00	0.17	0.00	NA	NA	0.01
Lubricant	0.00	0.00	0.00	0.00	5.01	0.00	NA	NA	0.31
Paint	31.9	24.5	81.0	90.1	73.6	14.1	NA	NA	41.51
Plastic	1.29	2.96	0.00	0.00	4.67	70.16	NA	NA	23.01
Polish	0.00	0.00	0.00	0.67	2.21	0.00	NA	NA	0.26
Speciality paper	0.00	3.98	0.00	0.00	0.00	0.00	NA	NA	0.75
Sporting Good	0.06	0.00	0.01	0.00	0.00	0.00	NA	NA	0.01
Textile	0.62	0.00	0.40	0.00	0.31	0.00	NA	NA	0.16
(2) nano-Al2O3									
Product categories	**WWI**	**B2B**	**BUND**	**Nanodb**	**Patent**	**Market method**	**Future market**	**Production volume**	**Weighted mean (%)**
DOB	**4**	**4**	**-**	**-**	**2**	**5**	**5**		
Adhesive	0.00	0.00	NA	NA	0.00	0.00	0.00	NA	0.00
Automotive coating	0.00	4.67	NA	NA	0.00	22.19	0.00	NA	6.48
Catalyst	0.00	0.74	NA	NA	0.83	0.00	25.0	NA	6.48

Cement	0.00	0.00	NA	NA	19.5	39.48	0.00	NA	11.8
Ceramic material	0.00	0.00	NA	NA	19.5	0.00	0.00	NA	1.95
Cleaning agent	40.9	0.00	NA	NA	0.00	0.00	0.00	NA	8.18
Cosmetics & personal care	34.6	0.00	NA	NA	1.22	0.68	10.0	NA	9.72
Electronics	0.00	0.00	NA	NA	0.08	0.00	20.0	NA	5.01
Filter	0.00	50.4	NA	NA	36.6	0.07	20.0	NA	18.8
Health Service	0.00	0.00	NA	NA	0.00	0.00	0.00	NA	0.00
Paint	0.00	6.49	NA	NA	0.00	36.6	25.0	NA	16.7
Plastic	0.00	0.26	NA	NA	0.22	0.98	0.00	NA	0.32
Polish	22.9	1.43	NA	NA	1.62	0.00	0.00	NA	5.04
Speciality paper	0.00	36.0	NA	NA	20.3	0.00	0.00	NA	9.23
Sporting Good	1.52	0.00	NA	NA	0.00	0.00	0.00	NA	0.30

(3) nano-FeOX

Category	**WWI**	**B2B**	**BUND**	**Nanodb**	**Patent**	**Market method**	**Future market**	**Produc-tion volume**	**Weighted mean (%)**
DOB	-	-	-	-	2	5	5	5	
Adhesive	NA	NA	NA	NA	0.04	0.00	0.00	0.00	2.77
Battery & Electrode	NA	NA	NA	NA	0.07	0.00	0.00	0.00	4.36
Catalysts	NA	NA	NA	NA	0.00	0.00	0.13	0.01	0.14
Ceramic	NA	NA	NA	NA	0.17	0.00	0.00	0.01	11.30
Paint	NA	NA	NA	NA	0.42	0.33	0.34	0.52	38.64
Constructionmaterial	NA	NA	NA	NA	0.28	0.58	0.00	0.32	38.02
Cosmetics	NA	NA	NA	NA	0.00	0.06	0.13	0.01	2.15
Electronics and Optics	NA	NA	NA	NA	0.00	0.03	0.28	0.04	1.00
Food Additives	NA	NA	NA	NA	0.00	0.00	0.00	0.01	0.00
Foundry sand	NA	NA	NA	NA	0.00	0.00	0.00	0.02	0.00
Medical	NA	NA	NA	NA	0.00	0.00	0.13	0.00	0.07
Plastics	NA	NA	NA	NA	0.00	0.00	0.00	0.05	0.08
Speciality Paper	NA	NA	NA	NA	0.00	0.00	0.00	0.02	0.00
Water treatment	NA	NA	NA	NA	0.02	0.00	0.00	0.00	1.52

(4) nano-CeO2

Product categories	**WWI**	**B2B**	**BUND**	**Nanodb**	**Patent**	**Market method**	**Future market**	**Produc-tion volume**	**Weighted mean (%)**
DOB	-	-	-	-	2	5	5	-	
Adhesive	NA	NA	NA	NA	0.01	0.00	0.00	NA	0.00
Composite	NA	NA	NA	NA	1.27	0.00	0.00	NA	0.21
Electric and Electronic	NA	NA	NA	NA	2.58	0.00	0.00	NA	0.43
Filter	NA	NA	NA	NA	0.01	0.00	0.00	NA	0.00
Fuel additive	NA	NA	NA	NA	10.48	0.28	35.00	NA	16.45
Fuel cell	NA	NA	NA	NA	4.59	18.12	0.00	NA	8.31
Glass	NA	NA	NA	NA	0.06	31.10	0.00	NA	12.97
Lubricant	NA	NA	NA	NA	0.06	0.00	0.00	NA	0.01
Medical	NA	NA	NA	NA	8.22	0.12	5.00	NA	3.51
Metal coating	NA	NA	NA	NA	0.59	10.66	0.00	NA	4.54
Other Catalyst	NA	NA	NA	NA	3.60	0.00	0.00	NA	0.60
Paint	NA	NA	NA	NA	17.62	0.00	15.00	NA	9.19
Personal care	NA	NA	NA	NA	0.00	4.35	0.00	NA	1.81
Polish	NA	NA	NA	NA	50.91	35.36	45.00	NA	41.97

(5) Quantum Dots

Product categories	**WWI**	**B2B**	**BUND**	**Nanodb**	**Patent**	**Market method**	**Allied market**	**Produc-tion volume**	**Weighted mean (%)**
DOB	-	-	-	-	2	-	5	-	
QD Catalyst	NA	NA	NA	NA	0.0342	NA	0.00	NA	0.01
QD Chips	NA	NA	NA	NA	0.1096	NA	0.09	NA	0.1
QD Display devices	NA	NA	NA	NA	0.2123	NA	0.21	NA	0.21
QD Ink	NA	NA	NA	NA	0.0411	NA	0.00	NA	0.01
QD Laser devices	NA	NA	NA	NA	0.0274	NA	0.06	NA	0.05
QD LED	NA	NA	NA	NA	0.2123	NA	0.09	NA	0.13
QD Medical devices	NA	NA	NA	NA	0.1301	NA	0.32	NA	0.27

QD Optical device	NA	NA	NA	NA	0.0411	NA	0.00	NA	0.01
QD Paint & Coating	NA	NA	NA	NA	0.0205	NA	0.00	NA	0.01
QD Photovoltaic devices	NA	NA	NA	NA	0.1164	NA	0.14	NA	0.13
QD Sensors	NA	NA	NA	NA	0.0548	NA	0.09	NA	0.08

NA: not available

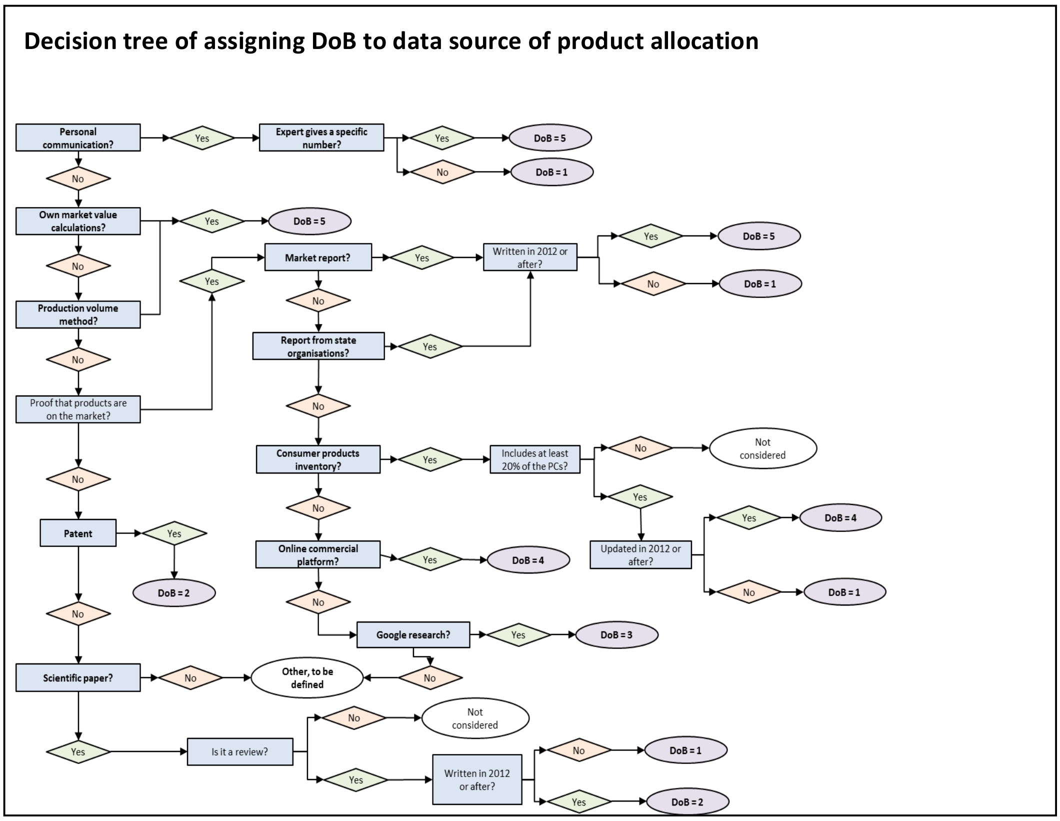


Figure S3.2. Decision tree of assigning degree of belief (DoB) to data sources of production of product allocation

4. Release schedule

Table S3.4. Release schedule and distribution of five ENMs

(a) nano- SiO2

Nano-SiO2	**Use release**	**Use release duration**	**Use release schedule**				**Use release distribution**				**EoL release**	**Lifetime distribution**	**EoL release distribution**				**Reference**
Product category	x		Y1	Y2	Y…	Yn	Waste water	Air	Surface water	Soil	1-X		Landfill	WIP	Recycling	Export	
Adhesive	0.01 (Sun et al. 2016)	2(Sun et al. 2016)	0.9	0.1			1				1	mean=8, 3σ=4 (JNIES 2016)	0.09	0.06	0.65	0.2	(Sun et al. 2016)
Automotive coating	0.01 (Sun et al. 2016)	1 *	1				0.8	0.1	0.1		0.99	mean=10, 3σ=5 (JNIES 2016)		0.1	0.9		(Institute 2014)
Car tire	0.13 (Würth 2007)	4 (Würth 2007)	1/4						0.0012	1	0.87	mean=4, 3σ=2 (Würth 2007)		0.038	0.642	0.32	(Würth 2007)
Catalyst	0.01[&]	1[&]	1				1				0.99	Y1=1.0			1		&
Ceramics	0.01 (Sun et al. 2016)	40 (JNIES 2016)	0.9	0.1/39			0.1		0.9		0.99	mean=40, 3σ=10 (JNIES 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Cleaning agent	0.95 (Sun et al. 2016)	1(Sun et al. 2016)	1				1				0.05	Y1=1.0 (Sun et al. 2016)	0.47[#]	0.37[#]	0.16[#]		(Eurostat 2017)
Cement	0.01 (Sun et al. 2016)	80 (Sun et al. 2016)	0.9	0.1/79			0.9			0.1	0.99	mean=80, 3σ=20 (Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Cosmetics & Personal care	0.95 (Sun et al. 2016)	2(Sun et al. 2016)	0.9	0.1			0.9		0.1		0.05	Y1=0.9, Y2=0.1(Sun et al. 2016)	0.47[#]	0.37[#]	0.16[#]		(Eurostat 2017)
Dental	0[&]	-									1	mean=8, 3σ=4 (Tomasi et al. 2008)		1			(ICRC 2011)
Electrical devices	0[&]	-									1	mean=8, 3σ=4 (JNIES 2016)	0.09	0.06	0.65	0.2	(Sun et al. 2016)
Food & Supplement	0.9 (Sun et al. 2016)	1(Sun et al. 2016)	1				1				0.1	Y1=1.0 (Sun et al. 2016)	0.6	0.4			(Sun et al. 2016)
Glass	0.01 (Sun et al. 2016)	80 (Sun et al. 2016)	0.9	0.1/79							0.99	mean=80, 3σ=20 (Sun et al. 2016)	0.97		0.03		(GoE 2013)
Ink	0.005[&]	5(Sun et al. 2016)	1/5					1			0.995	mean=5, 3σ=4 (Sun et al. 2016)	0.07	0.03	0.7	0.2	(Sun et al. 2016)
Lubricant	0.05[&]	2[&]	0.9	0.1			1				0.95	Y1=0.9, Y2=0.1 (Sun et al. 2016)		1			&
Paint	0.01 (Sun et al. 2016)	7 (Sun et al. 2016)	0.9	0.1/6			0.5	0.25		0.25	0.99	mean=80, 3σ=20 (Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Plastic	0.015 (Sun et al. 2016)	3[&]	1/3				1				0.985	mean=3, 3σ=2 [&]	0.47[#]	0.37[#]	0.16[#]		(Eurostat 2017)
Polish	0.95 (Sun et al. 2016)	2 (Sun et al. 2016)	0.9	0.1				0.5	0.2	0.3	0.05	Y1=0.9, Y2=0.1 (Sun et al. 2016)	0.47[#]	0.37[#]	0.16[#]		(Eurostat 2017)
Speciality paper	0 (Sun et al. 2016)	-									1	mean=5, 3σ=4 (Sun et al. 2016)	0.07	0.03	0.7	0.2	(Sun et al. 2016)
Sporting Good	0.04 (Sun et al. 2016)	7 (JNIES 2016)	1/7				0.7	0.3			0.96	mean=7, 3σ=3 (JNIES 2016)	0.47[#]	0.37[#]	0.16[#]		(Eurostat 2017)
Textiles	0.6 (Sun et al. 2016)	3(Sun et al. 2016)	0.7	0.2	0.1		0.8	0.2			0.4	mean=3, 3σ=2 (Sun et al. 2016)	0.31	0.07	0.28	0.34	(Sun et al. 2016)

(b) nano-CeO2

Nano-CeO2	**Use release**	**Use release duration**	**Use release schedule**	**Use release distribution**	**EoL release**	**Lifetime distribution**	**EoL release distribution**	**Reference**

Product Category	x		Y1	Y2	Y…	Yn	Waste water	Air	Surface water	Soil	1-x		Landfill	WIP	Recycling	Export	
Adhesive	0.01 (Sun et al. 2016)	2&	0.9	0.1			1				0.99	Y1=0.9, Y2=0.1&	0.47#	0.37#	0.16#		(Eurostat 2017)
Composite	0.05 (Sun et al. 2016)	5&	1/5								0.95	mean=5, 3σ=2&	0.47#	0.37#	0.16#		(Eurostat 2017)
Electric and Electronic	0&							1			1	mean=6, 3σ=4 (JNIES 2016)	0.1	0.05	0.65	0.2	(Sun et al. 2016)
Filter	0.05&	4&	0.05*(1/4)				1				0.95	mean=4, 3σ=1&	0.47#	0.37#	0.16#		(Eurostat 2017)
Fuel additive	0.01 (ATC 2013)	1&	1					1			0.99	mean=3, 3σ=1&			1		(Collin et al. 2014)
Fuel cell	0&	0									1	mean=10, 3σ=5 (JNIES 2016)			1		&
Glass	0&										1	mean=80, 3σ=20 (Sun et al. 2016)	1				(GoE 2013)
Lubricant	0.05&	1&	0.9	0.1			1				0.95	Y1=0.9, Y2=0.1 (Sun et al. 2016)		1			&
Medical	1&	1&	1				1				0	0					(ICRC 2011)
Metal coating	0.05 (Sun et al. 2016)	40 (Sun et al. 2016)	0.9	0.1*(1/39)			1				0.95	mean=20, 3σ=5 (Sun et al. 2016)	0.03	0.02	0.95		(Sun et al. 2016)
Catalyst	0.01 (Sun et al. 2016)	1&	1				1				0.99	1&			1		&
Paint	0.05 (Sun et al. 2016)	7 (Sun et al. 2016)	0.9	0.1*(1/6)			0.9		0.1		0.95	mean=80, 3σ=20 (Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Personal care	0.95 (Sun et al. 2016)	2 (Sun et al. 2016)	0.9	0.1			0.9		0.1		0.05	Y1=0.9, Y2=0.1 (Sun et al. 2016)	0.47#	0.37#	0.16#		(Sun et al. 2016)
Polish	0.1@	2&	0.9	0.1			0.99	0.01 (Choi et al. 2015)			0.9	Y1=0.9, Y2=0.1&	0.9		0.1		(Jack Hsu et al. 2010)

@ Personal communication with expert from HighQfactory GmbH

(c) nano- Al$_2$O$_3$															
nano- Al2O3	**Use release**	**Use Release duration**	**Use release schedule**		**Use release distribution**				**EoL release**	**Lifetime distribution**	**EoL release distribution**				**Reference**
Product Category	x		Y1	Y2	Waste water	Air	Surface water	Soil	1-x		Landfill	WIP	Recycling	Export	
Adhesive	0.05 (Sun et al. 2016)	2 &	0.9	0.1	1				0.95	mean =40, 3σ=10(JNIES 2016)	0.139	0.0006	0.8604		(Eurostat 2014)
Automotive coating	0.01 (Sun et al. 2016)	2 (Sun et al. 2016)	0.9	0.1	0.8	0.1	0.1		0.99	mean=10, 3σ=5(JNIES 2016)		0.1	0.9		(Institute 2014)
Catalyst	0.01&	1 (Sun et al. 2016)	1		1				0.99	1&			1		&
Cement	0.01 (Sun et al. 2016)	80 (Sun et al. 2016)	0.9	0.1/79	0.9		0.1		0.99	mean=80, 3σ=20(Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Ceramic material	0.01 (Sun et al. 2016)	40 (Sun et al. 2016)	0.9	0.1/39	0.9		0.1		0.99	mean=40, 3σ=10(JNIES 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Cleaning agent	0.95 (Sun et al. 2016)	1 (Sun et al. 2016)	1		1				0.05	Y1=1.0(Sun et al. 2016)	0.47	0.37	0.16		(Eurostat 2017)
Cosmetics & personal care	0.95 (Sun et al. 2016)	2 (Sun et al. 2016)	0.9	0.1	0.9		0.1		0.05	Y1=0.9, Y2=0.1	0.47#	0.37#	0.16#		(Eurostat 2017)
Electronics	0&	6&	1/6						1	mean=6, 3σ=4(JNIES 2016)	0.09	0.06	0.65	0.2	(Sun et al. 2016)
Filter	0.05&	2&	0.5	0.5	1				0.95	mean=2, 3σ=2&	0.47#	0.37#	0.16#		(Eurostat 2017)
Health Service	0&	8&							1	mean=8, 3σ=2&		1			(ICRC 2011)

Paint	0.05 (Sun et al. 2016)	7	0.9	0.1*(1/6)	0.9		0.1		0.95	mean =80, 3σ=20(Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Plastic	0.015 (Sun et al. 2016)	3	1/3		1				0.985	mean=3, 3σ=2(Sun et al. 2016)	0.47#	0.37#	0.16#		(Eurostat 2017)
Polish	0.95 (Sun et al. 2016)	2&	0.9	0.1	0.5	0.2	0.3		0.05	Y1=0.9, Y2=0.1(Sun et al. 2016)	0.47#	0.37#	0.16#		(Eurostat 2017)
Speciality paper	0 (Sun et al. 2016)								1	mean=5, 3σ=4(Sun et al. 2016)	0.07	0.03	0.7	0.2	(Sun et al. 2016)
Sporting Good	0.04 (Sun et al. 2016)	7 (Sun et al. 2016)	0.4*(1/7)						0.96	mean=7, 3σ=3(Sun et al. 2016)	0.47#	0.37#	0.16#		(Eurostat 2017)

(d) nano- FeOX

Nano Iron oxide	**Use release**	**Use release duration**	**Use release schedule**		**Use release distribution**				**EoL release**	**Lifetime distribution**	**EoL release distribution**				**Reference**
Product Category	**x**		**Y1**	**Y2**	**Waste water**	**Air**	**Surface water**	**Soil**	**1-x**		**Landfill**	**WIP**	**Recycling**	**Export**	
Adhesive	0.01(Sun et al. 2016)	2	0.9	0.1	1				0.99	Y1=0.9, Y2=0.1	0.47#	0.37#	0.16#		(Eurostat 2017)
Battery & Electrode	0(Sun et al. 2016)	11							1	mean= 11, 3σ=5	0.45	0.3	0.25		(Sun et al. 2016)
Catalysts	0.05&	1	1		1				0.95	1			1		&
Ceramic	0.01(Sun et al. 2016)	40	0.9	0.1/39	0.9		0.1		0.99	mean=40, 3σ=10(JNIES 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Paint	0.01(Sun et al. 2016)	7	0.9	0.1*(1/6)	0.9		0.1		0.99	mean= 80, 3σ=20(Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Cement	0.01(Sun et al. 2016)	80	0.9	0.1*(1/79)	0.9		0.1			mean=80, 3σ=20(Sun et al. 2016)	0.139*	0.0006*	0.8604*		(Eurostat 2014)
Cosmetics	0.95(Sun et al. 2016)	2	0.9	0.1	0.9		0.1		0.05	Y1=0.9, Y2=0.1(Sun et al. 2016)	0.47#	0.37#	0.16#		(Eurostat 2017)
Electronics	0&	13	1/13						1	mean=13, 3σ=8(JNIES 2016)	0.09	0.06	0.65	0.2	(Sun et al. 2016)
Food Additives	0.95 &	1&	1		1				0.05	Y1=1(Sun et al. 2016)	0.6	0.4			(Sun et al. 2016)
Foundry sand	1&	1						1	0	--					(EPA 2008)
Medical	0.95 &	3	1/3		1				0.05	mean=3, 3σ=2&		1			(ICRC 2011)
Plastics	0.015(Sun et al. 2016)	3	1/3		1				0.985	mean=3, 3σ=2(Sun et al. 2016)	0.47#	0.37#	0.16#		(Eurostat 2014)
Speciality Paper	0(Sun et al. 2016)	2							1	mean=5, 3σ=4(Sun et al. 2016)	0.07	0.03	0.7	0.2	(Sun et al. 2016)
Water treatment	0.95&	1	1				0.9	0.1	0.05	Y1=1&	1				&

& Expert evaluation

* Mineral waste from construction and demolition handling. The data present here is for the EU. Data for other regions are presented in Table S5.

Mixed waste from household management. The data present here is for the EU. Data for other regions are presented in Table S5.

Table S5. Wastewater, waste, and sludge treatment in seven regions

<table>
<tr><th colspan="8">Average percentage of wastewater treated in different regions (EEA. 2013)</th></tr>
<tr><td>Regions →</td><td>EU</td><td>CH</td><td>CE</td><td>EE</td><td>NE</td><td>SE</td><td>SEE</td></tr>
<tr><td>Average percentage of wastewater treated (%)</td><td>86.7</td><td>97.7-98</td><td>96.7</td><td>78.8</td><td>92.5</td><td>82.8</td><td>63.3</td></tr>
<tr><th colspan="8">Average percentage of mixed waste in each treatment method in different regions (%) (Eurostat 2017)</th></tr>
<tr><td>Treatment methods ↓ \ Regions →</td><td>EU</td><td>CH</td><td>CE</td><td>EE</td><td>NE</td><td>SE</td><td>SEE</td></tr>
<tr><td>Landfilling</td><td>46.9</td><td>0</td><td>20.7</td><td>74.7</td><td>27.5</td><td>72.7</td><td>94.6</td></tr>
<tr><td>Incineration</td><td>37</td><td>100</td><td>68.2</td><td>13.4</td><td>51.6</td><td>13.8</td><td>2.47</td></tr>
<tr><td>Recovering</td><td>16.1</td><td>0</td><td>11.1</td><td>11.9</td><td>20.9</td><td>13.5</td><td>2.93</td></tr>
<tr><th colspan="8">Average percentage of construction waste in each treatment method in different regions (%) (Eurostat 2014)</th></tr>
<tr><td>Treatment method ↓ \ Regions →</td><td>EU</td><td>CH</td><td>CE</td><td>EE</td><td>NE</td><td>SE</td><td>SEE</td></tr>
<tr><td>Landfilling</td><td>13.9</td><td>23</td><td>5.6</td><td>16.3</td><td>30.8</td><td>42.6</td><td>66.0</td></tr>
<tr><td>Incineration</td><td>86</td><td>77</td><td>93.4</td><td>83.5</td><td>66.3</td><td>57.4</td><td>33.4</td></tr>
<tr><td>Recovering</td><td>0.1</td><td>0</td><td>0.95</td><td>0.20</td><td>2.70</td><td>0.0086</td><td>0.70</td></tr>
<tr><td>Export</td><td>0</td><td>0</td><td>0.05</td><td>0</td><td>0</td><td>0</td><td>0</td></tr>
<tr><th colspan="8">Average percentage of STP sludge in each treatment method in different regions (%) (Eurostat 2016)</th></tr>
<tr><td>Treatment method ↓ \ Regions →</td><td>EU</td><td>CH</td><td>CE</td><td>EE</td><td>NE</td><td>SE</td><td>SEE</td></tr>
<tr><td>Landfilling</td><td>21</td><td>0</td><td>4.4</td><td>47.6</td><td>23.2</td><td>28</td><td>90.6</td></tr>
<tr><td>Incineration</td><td>24</td><td>100</td><td>49.6</td><td>13.8</td><td>7.1</td><td>7.3</td><td>3.94</td></tr>
<tr><td>Agriculture application</td><td>55</td><td>0</td><td>46</td><td>38.6</td><td>69.7</td><td>64.7</td><td>5.46</td></tr>
</table>

EU: EU 27, CH: Switzerland, CE: Central Europe, EE: Eastern Europe, SE: Southern Europe, NE: Northern Europe, SEE: South-eastern Europe

Table S3.6. Concentrations of five ENMs in different environmental compartments and waste management processes in seven systems in 2014

nano-SiO2		EU				Central Europe				Northern Europe				Eastern Europe				Southern Europe				Southeastern Europe				Switzerland				
		Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Unit
	Effluent	65	28	1.6	140	74	32	1.6	15	48	19	0.97	100	51	19	0.95	110	23	6.9	0.43	48.9	34	13	0.6	72	12	4.8	0.21	25	µg/L
	Sludge	4.4	3.6	0.15	8.5	5.3	4.8	0.16	10	4.6	4	0.14	8.9	2.4	1.8	0.06	4.7	2.1	0.96	0.06	4.3	3.3	2.6	0.09	6.6	6.8	5.3	0.17	14	g/kg
	Landfilled waste	490	380	130	860	620	490	160	1000	1500	1200	380	2700	450	350	110	770	400	310	83	730	200	150	47	360	-	-	-	-	mg/kg
	Incinerated waste	490	400	140	850	620	500	180	1000	470	360	120	810	380	300	120	660	180	140	49	310	5900	4800	1900	9800	670	540	170	1200	mg/kg
	Bottom ash	3.4	2.8	1.20	5.7	4.5	3.9	1.4	7.4	3.1	2.5	0.98	5.2	2.8	2.4	0.9	4.8	1.1	0.96	0.34	1.9	25	22	9.3	42	5.7	5.1	1.7	9.3	g/kg
	Flyash	4.7	3.8	1.50	7.9	6.1	5	1.8	10	4.1	3.4	1.3	7.1	3.9	3.2	1.1	6.6	1.5	1.3	0.46	2.6	35	29	12	58	7.9	6.9	2.2	13	g/kg
	Air	16	13	0.33	3.1	34	30	0.73	66	2.6	2.2	0.05	5.1	8.7	6.6	0.15	18	10	4.4	0.19	22	18	14	0.31	36	48	34	0.68	100	ng/m3
	NU_Soil	86	80	49	120	170	160	99	240	28	25	15	40	52	47	30	75	68	61	37	99	92	84	51	130	290	270	160	430	µg/kg
	ST Soil	150	140	84	220	390	350	200	350	240	230	140	350	330	300	175	490	240	220	130	360	110	110	60	170	-	-	-	-	mg/kg
	Surface water	4.3	3.5	0.2	8.5	4.4	3.5	0.14	8.8	0.22	0.18	0.01	0.45	2.5	1.8	0.07	5.1	8.6	3.9	0.2	18	11	9	0.3	23	4.2	3	0.11	8.8	µg/L
	Sediments	79	73	44	114	79	70	42	110	4.1	3.7	2.2	6	46	42	25	66	180	160	90	260	210	190	110	302	75	68	34	110	mg/kg
nano-iron oxide		EU				Central Europe				Northern Europe				Eastern Europe				Southern Europe				Southeastern Europe				Switzerland				
		Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Unit
	Effluent	1.5	0.2	0.05	1.8	1.3	0.23	0.05	1.6	0.85	0.17	0.03	1.2	1.2	0.16	0.03	1.4	0.7	0.11	0.02	0.77	0.56	0.08	0.01	0.68	0.24	0.05	0.01	0.33	µg/L
	Sludge	75	19	5.1	130	85	21	5.9	140	73	18	4.4	120	44	8.6	2.3	68	53	11	3.1	79	45	10	1.9	68	110	35	5.8	200	mg/kg
	Landfilled waste	0.89	0.46	0.17	1.5	1.9	1.1	0.42	3.2	4.7	2.7	1.0	7.8	1.03	0.43	0.15	1.8	1.2	0.51	0.19	1.8	0.35	0.15	0.053	0.62	-	-	-	-	mg/kg
	Incinerated waste	1.0	0.51	0.19	1.8	1.2	0.52	0.20	2.1	0.94	0.45	0.16	1.6	1.3	0.7	0.24	2.3	0.67	0.37	0.14	1.1	19	11	3.9	31	1.4	0.81	0.23	2.4	mg/kg
	Bottom ash	22	9.8	3.7	36	31	11	4.2	50	11	6.1	2.3	18	32	15	5.2	50	14	6.9	2.7	21	240	140	49	420	42	17	5.5	72	mg/kg
	Flyash	32	14	5.0	50	44	16	5.7	66	16	8.4	3.1	26	44	21	7.0	69	19	9.6	3.6	31	340	190	67	560	59	24	7.4	96	mg/kg
	Air	0.23	0.04	0.0062	0.3	0.53	0.086	0.017	0.74	0.039	0.0063	0.0011	0.054	0.15	0.020	0.0031	0.21	0.25	0.037	0.0058	0.32	0.24	0.03	0	0.34	0.76	0.13	0.012	1.4	ng/m3
	NU_Soil	0.32	0.21	0.09	0.58	0.67	0.44	0.17	1.2	0.11	0.074	0.031	0.19	0.22	0.14	0.053	0.39	0.33	0.22	0.086	0.56	0.31	0.2	0.08	0.53	1.2	0.88	0.42	1.9	µg/kg
	ST Soil	2.7	1.9	0.76	4.9	6.6	4.3	1.7	12	4.2	2.9	1.2	7.6	6.0	3.8	1.4	11	5.8	3.9	1.6	11	1.7	1.1	0.43	3.1	-	-	-	-	mg/kg
	Surface water	86	17	0.41	110	2.0	1.32	0.52	3.6	4.7	1.1	0.25	7.4	58	10	208	91	250	51	13	350	180	37	7.3	0.27	110	27	3.8	190	ng/L
	Sediments	1.6	0.87	0.35	2.3	85	21	5.9	140	0.094	0.065	0.026	0.17	1.03	0.67	0.24	1.8	4.4	3.0	1.2	7.7	3.6	3.4	0.92	6.4	2.0	1.5	0.70	3.3	mg/kg
nano-CeO2		EU				Central Europe				Northern Europe				Eastern Europe				Southern Europe				Southeastern Europe				Switzerland				
		Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Mean	Median	Q15	Q85	Unit
	Effluent	37	11	1.3	66	44	19	1.5	77	29	7.7	0.93	55	25	7.7	1.1	44	19	4.8	0.64	34	18	2.8	0.48	35	4.8	1.3	0.13	9.1	ng/l
	Sludge	1.8	0.55	0.07	3.7	2.5	1.2	0.09	4.8	2	0.61	0.07	4.3	0.89	0.29	0.04	1.7	1.2	0.34	0.04	2.4	1.3	0.22	0.04	2.9	2.6	0.93	0.1	5.4	mg/kg
	Landfilled waste	6.5	2.4	0.27	13	23	12	1.1	43	41	14	1.5	81	3.3	1.2	0.20	6.3	4.4	1.5	0.17	8.8	1.3	0.27	0.03	2.9	-	-	-	-	mg/kg
	Incinerated waste	8.4	5.5	1.9	15	10	7.4	2.8	18	7.8	5.1	1.8	14	8.1	4.8	2.0	14	4.7	2.8	1.1	8.3	150	87	27	270	10	6.7	2.3	18	µg/kg
	Bottom ash	0.55	0.2	0.05	1.1	0.85	0.41	0.07	1.5	0.25	0.11	0.03	0.49	0.66	0.24	0.07	1.2	0.32	0.11	0.03	0.61	7.1	1.7	0.64	14	0.9	0.36	0.07	1.8	mg/kg
	Flyash	0.77	0.28	0.07	1.5	1.2	0.54	0.09	2.2	0.36	0.15	0.05	0.66	0.91	0.35	0.10	1.5	0.45	0.15	0.04	0.83	9.8	2.5	0.86	19	1.2	0.45	0.1	2.4	mg/kg
	Air	42	11	0.6	86	100	47	1.4	210	6.8	1.7	0.1	14	20	5.8	0.22	37	35	8.9	0.53	73	43	5.3	0.57	97	110	36	1.6	230	pg/m3
	NU_Soil	42	35	17	69	96	53	45	140	14	12	5.9	23	22	18	9.1	36	39	33	17	61	41	33	16	68	120	103	54	190	ng/kg
	ST Soil	60	50	26	95	180	150	83	270	110	93	46	170	120	96	48	###	130	110	54	200	45	37	17	74	-	-	-	-	µg/kg
	Surface water	2	0.6	0.08	4.1	2.6	1.3	0.11	4.9	0.11	0.03	0	0.23	1.1	0.35	0.54	20	5.1	1.4	0.19	10	4.9	0.78	0.14	11	1.9	0.65	0.07	3.7	ng/l
	Sediments	35	29	15	57	46	39	21	71	2.1	1.8	0.9	3.4	19	16	8.1	32	95	81	40	150	87	71	34	140	32	27	14	50	µg/kg

		EU				Central Europe				Northern Europe				Eastern Europe				Southern Europe				Southeastern Europe				Switzerland				
		Mean	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Unit**
nano-Al2O3	**Effluent**	3.6	0.96	0.09	5.6	5.1	1.5	0.15	8.4	2.6	0.67	0.08	4.1	3.5	1.1	0.11	5.9	2.5	0.98	0.14	3.9	2.5	0.75	0.06	4.3	1.1	0.28	0.03	1.4	**µg/L**
	Sludge	240	110	7.2	330	380	190	17	550	250	89	9.9	390	160	99	8.1	250	220	140	19	310	240	130	8.2	360	620	270	19	750	**mg/kg**
	Landfilled waste	9.9	5.6	2.2	14	20	13	4.9	31	41	24	8.6	58	13	8.5	2.9	20	16	11	4.5	25	4.7	3.0	1.0	7.7	-	-	-	-	**mg/kg**
	Incinerated waste	9.6	5.6	2.3	13	17	11	4.1	26	9.9	6.0	2.1	14	11	7.7	3.2	18	6.1	4.5	2.1	9.6	140	100	48	220	21	11	3.9	33	**mg/kg**
	Bottom ash	93	60	21	120	190	120	38	270	64	40	16	88	120	87	32	180	55	41	20	84	870	630	320	1300	300	150	45	460	**mg/kg**
	Flyash	130	82	27	170	260	160	55	400	88	56	20	120	170	120	43	260	77	56	26	120	1200	870	420	1800	410	210	62	640	**mg/kg**
	Air	1.2	0.57	0.038	1.7	3.7	1.7	0.16	5.6	0.21	0.076	0.0088	0.33	0.88	0.49	0.045	1.4	1.6	1.0	0.14	2.3	1.9	1.0	0.067	2.9	6.5	2.8	2.2	7.9	**ng/m3**
	NU_Soil	1.3	0.93	0.50	1.9	4.1	3.2	1.8	5.9	0.51	0.38	0.22	0.77	160	99	8.1	250	1.9	1.6	0.96	2.9	2.0	1.6	0.94	3.1	8.4	6.2	3.3	14	**µg/kg**
	ST Soil	8.2	5.9	3.5	12	27	22	12	41	14	9.9	6.0	20	1.2	1.0	0.55	1.8	23	19	11	33	7.9	6.3	3.4	12	-	-	-	-	**mg/kg**
	Surface water	0.31	0.15	0.0096	0.43	0.53	0.25	0.02	0.78	0.02	0.01	0	0.03	0.22	0.12	0.01	0.35	1.1	0.71	0.1	1.5	1.0	0.54	0.033	1.5	0.65	0.28	0.02	0.81	**µg/L**
	Sediments	5.7	4.1	2.3	8.5	9.3	7.5	4.2	14	0.33	0.24	0.14	0.48	4.1	3.4	2.0	6.1	19	16	9.8	29	17	14	7.8	26	11	8.0	4.2	18	**mg/kg**
		EU				**Central Europe**				**Northern Europe**				**Eastern Europe**				**Southern Europe**				**Southeastern Europe**				**Switzerland**				
		Mean	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Mean**	**Median**	**Q15**	**Q85**	**Unit**
Quantum dots	**Effluent**	2.7	1.5	4.9	5.1	3.2	1.6	0.52	6	2.3	1.3	0.39	4.2	2.4	1.3	0.41	4.5	1.6	0.81	0.24	3.0	1.6	0.83	0.26	3.0	0.48	0.27	0.093	0.85	**pg/L**
	Sludge	0.18	0.11	0.06	0.37	0.23	0.14	0.07	0.49	0.22	0.13	0.076	0.46	110	72	38	230	0.14	0.083	0.043	0.29	0.15	0.088	0.050	0.32	0.26	0.16	0.09	0.53	**fg/L**
	Landfilled waste	91	82	49	140	330	290	170	480	660	600	360	970	57	52	32	84	70	62	37	10	22	19	11	32	-	-	-	-	**ng/kg**
	Incinerated waste	1.6	1.0	0.59	3.1	1.2	0.74	0.43	2.4	1.2	0.75	0.44	2.3	3.4	2.3	1.3	6.6	1.8	1.1	0.64	3.5	72	44	27	150	0.83	0.55	0.32	1.64	**µg/kg**
	Bottom ash	8.0	5.2	2.8	16	6.1	4.0	2.1	12	6.1	4.0	2.3	12	17	11	6.2	32	9.3	5.9	3.3	18	260	150	90	520	4.8	3.1	1.8	9.3	**pg/kg**
	Flyash	11	7.3	3.8	21	8.4	5.2	2.8	16	8.5	5.5	3.0	16	23	15	8.2	42	13	8.1	4.3	25	350	220	120	370	6.6	4.3	2.3	13	**ng/kg**
	Air	7.9	4.8	2.5	1.6	19	11	5.8	3.8	1.5	0.93	0.50	3.0	5.1	3.1	1.6	10	8.4	4.9	2.6	17	10	5.8	3.3	22	22	13	7.5	43	**fg/m3**
	NU_Soil	8.4	7.7	5.1	12	18	16	11	25	3.0	2.7	1.8	4.3	5.8	5.4	3.4	8.1	9.1	8.5	5.5	13	9.9	9.3	5.9	14	27	25	17	38	**ng/kg**
	ST Soil	6.2	5.8	3.8	8.7	17	15	9.5	24	11	10	6.8	16	16	15	8.8	23	15	13	9.1	20	5.3	4.9	3.0	7.4	-	-	-	-	**µg/kg**
	Surface water	170	150	56	350	180	100	55	370	9.6	5.9	3.2	19	120	71	37	220	500	290	150	1000	0.53	0.30	0.16	1.1	150	93	49	290	**µg/kg**
	Sediments	3.2	2.9	1.9	4.3	3.2	2.9	1.8	4.5	0.17	0.15	0.098	0.24	2.1	1.9	1.2	2.9	9.1	8.5	5.4	13	9.4	8.8	5.6	13	2.8	2.6	1.6	4.0	**µg/kg**

Table S3.7. Volume and mass of the receiving media in waste handling process and environmental compartments

Summary of volumes of different technical and environmental compartments used for Central Europe (ECHA 2016)								
Compartments ↓ \ Regions →	**EU**	**CE**	**EE**	**NE**	**SE**	**SEE**	**CH**	**Units**
Air	4.33E+15	9.17E+14	7.3E+14	1.3E+15	1.59E+15	3.73E+14	4.13E+13	m^3
Natural and urban soil	7.59E+14	1.71E+14	1.2E+14	1.1E+14	2.71E+14	7.09E+13	6.25E+12	kg
Sludge treated soil	7.43E+11	1.31E+11	2.2E+10	3.3E+10	1.50E+11	7.53E+09	0.00E+00	kg
Surface water	3.89E+14	8.25E+13	6.6E+13	2.3E+14	4.78E+13	2.24E+13	2.02E+10	litre
Sediment	1.01E+12	2.14E+11	1.7E+11	6E+11	1.24E+11	5.82E+10	1.35E+10	kg
STP inffluent	2.97E+13	1.57E+13	3.5E+12	2.3E+12	2.10E+13	4.66E+12	5.83E+12	litre
STP sludge	9.00E+09	4.14E+09	1.5E+09	4.7E+08	4.65E+09	9.16E+08	2.03E+08	kg
Solid waste landfilled	7.31E+10	1.09E+10	1.2E+10	6.00E+08	3.66E+10	3.36E+10	0.00E+00	kg
Solid waste incinerated	6.16E+10	3.96E+10	3.1E+09	4.6E+09	2.10E+10	1.84E+08	2.85E+09	kg

EU: EU 27, CH: Switzerland, CE: Central Europe, EE: Eastern Europe, SE: Southern Europe, NE: Northern Europe, SEE: South-eastern Europe

5. *Reference*

Anonymous 2012: COMMISSION STAFF WORKING PAPER : Types and uses of nanomaterials, including safety aspect

ANSES 2013: Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)

ANSES (2014): Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude. ANSES (l'Agence nationale de sécurité sanitaire)

ANSES (2015): Éléments issus des déclarations des substances à l'état nanoparticulaire. RAPPORT d'étude.

ATC 2013: Fuel Additives: Use and Benefits

BUND 2011: Nanoproduktdatenbank Germany

Choi KM, Kim JH, Park JH, Kim KS, Bae GN (2015): Exposure Characteristics of Nanoparticles as Process By-products for the Semiconductor Manufacturing Industry. Journal of Occupational and Environmental Hygiene 12, D153-D160

Collin B, Oostveen E, Tsyusko OV, Unrine JM (2014): Influence of Natural Organic Matter and Surface Charge on the Toxicity and Bioaccumulation of Functionalized Ceria Nanoparticles in Caenorhabditis elegans. Environmental Science & Technology 48, 1280-1289

Commission E 2012: Types and uses of nanomaterials, including safety aspects

EC21 (2015): Global B2b Market Place Available from: http://manufacturer.ec21.com/nano_silica.html. Retrieved 15.03.2015.

ECHA 2016: Guidance on Information Requirements and Chemical Safety Assessment Chapter R.16: Environmental Exposure Assessment

EEA. EEA 2013: Urban Waste Water Treatment, Kongens Nytorv 6, 1050 Copenhagen K Denmark

EPA 2008: BENEFICIAL REUSE OF FOUNDRY SAND: A REVIEW OF STATE PRACTICES AND REGULATIONS, Washington, DC

EPR (2015): European Patent Office. European Patent Register Available from: http://www.epo.org/index.html. Retrieved 14.03.2015.

Eurostat (2014): Treatment of Waste by Waste Category, Hazardousness and Waste Operations-- Mineral Waste from Construction and Demolition

Eurostat (2016): Sewage sludge production and disposal.

Eurostat (2017): Municipal Waste Stastistic- mixed waste

Future Market (2014): Nanomaterials Producers Directory 2014-2015.

Future Markets 2012: The global market for nanomaterials 2002–2006: production volumes, revenues and end use markets. Future Markets Inc., http://www.futuremarketsinc. com/index.php?option=com_content&view=article&id= 176&Itemid=73

Future Markets (2014): The Global Market for Metal Oxide Nanoparticles 2014. Future Market

GoE 2013: Recycling of end-of-life building glass

ICRC 2011: Medical Waste Management

Institute SR 2014: RECYCLING SCRAPPED AUTOMOBILES

Jack Hsu, Y.J.Wann, J.R.Pai 2010: CMP Oxide Slurry recycle system implementation

JNIES (2016): Lifespan database for Vehicles, Equipment, and Structures: LiVES. Japanese National Institute for Environmetal Studies (JNIES)

Nanodatabase (2013): The Nanodatabase http://nanodb.dk/.

Piccinno F, Gottschalk F, Seeger S, Nowack B (2012): Industrial Production Quantities and Uses of Ten Engineered Nanomaterials for Europe and the World. J. Nanopart. Res. 14, 1109

Schmid K, Riediker M (2008): Use of nanoparticles in Swiss industry: A targeted survey. Environ. Sci. Technol. 42, 2253-2260

Shreyas Naidu, Gotsurve P 2014: Quantum Dots Market

Global Share, Size, Industry Analysis, Trends, Opportunities, Growth and Forecast, 2013 - 2020, Allied Market Reasearch

Sun TY, Bornhoft NA, Hungerbuhler K, Nowack B (2016): Dynamic Probabilistic Modeling of Environmental Emissions of Engineered Nanomaterials. Environ Sci Technol 50, 4701-11

Tomasi C, Wennstrom JL, Berglundh T (2008): Longevity of teeth and implants - a systematic review. J Oral Rehabil 35 Suppl 1, 23-32

USP&TO (2015): United States Patent and Trademark Office. Patent Full-Text Databases Available from: http://patft.uspto.gov/netahtml/PTO/index.html. Retrieved 14.03.2015.

Wang Y (2016): Survey on Production and application of five nanomaterials (nano-Al2O3, nano-SiO2, nano iron oxide, nano-CeO2 and quantum dots)

Wang Y, Deng L, Caballero-Guzman A, Nowack B (2016a): Are engineered nano iron oxide particles safe? an environmental risk assessment by probabilistic exposure, effects and risk modeling. Nanotoxicology 10, 1545-1554

Wang Y, Kalinina A, Sun T, Nowack B (2016b): Probabilistic modeling of the flows and environmental risks of nano-silica. Sci Total Environ 545-546, 67-76

Willcocks D (2007): NICNAS Information Sheet, Summary of call for information on the use of Nanomaterials; Australian Government.

Würth B 2007: Emission of engineered and unintentionally procuced nanoparticles to the soil, an exposure assessment for Switzerland.

WWI; (2012): The American Woodrow Wilson (WW) database WWI: The Project on Emerging Nanotechnology Inventory.

Supporting information for Annex 4

Table S4.1 Endpoint concentrations of nano-SiO_2, nano iron oxides, nano-Al_2O_3, nano-CeO_2, and quantum dots

Environmental compartment	**Taxonomic group**	**Test Organism**	**Particle Size (nm)**	**Test material**	**Test condition**	**Exposure time (h)**	**Test endpoint**	**Ecotoxic endpoint**	**Concentration (µg/l)**	**Assessment Factor (AF) for the extrapolation from short to long term effect**	**Assessment Factor (AF) for the extrapolation from the observed effect into no effect concentrations**	**Species sensitivity [µg/l]**	**Reference**
Water	Algae	Pseudokirchneriella subspicata	13	Colloidal SAS in suspensions.	25 ± 1 °C.	72	Cell density and average specific growth rate	EC20	20000	10	2	1000	[1]
Water	Algae	Pseudokirchneriella subspicata	27	Colloidal SAS in suspensions.	25 ± 1 °C.	72	Cell density and average specific growth rate	EC20	28800	10	2	1440	[1]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	pH= 7.6	48	Algal growth inhibition	NOEC	4600	10	1	460	[1]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	pH=8.6	48	Algal growth inhibition	NOEC	10000	10	1	1000	[2]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	NOM 1.2 mg C/l	48	Algal growth inhibition	NOEC	22000	10	1	2200	[2]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	NOM 4.7mg C/l	48	Algal growth inhibition	NOEC	100000	10	1	10000	[2]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	NOM 9mg C/l	48	Algal growth inhibition	NOEC	100000	10	1	10000	[2]
Water	Algae	Pseudokirchneriella subspicata	22	Colloidal SAS in suspensions.	pH=6.8	48	Algal growth inhibition	NOEC	100000	10	1	10000	[2]

Water	Algae	Pseudokirchneriella subspicata	14	Silica nanoparticle suspension	23.0 ± 1.2◦C and pH7.6 ± 0.4.	96	Chlorophyll content	EC20	318000	1	2	159000	[3]
Water	Algae	Pseudokirchneriella subcapitata	50 and 100	Green fluorescently labeled silica nanoparticles	20±1 °C	72	Growth	HNOEC	>100000	10	1	>10000	[4]
Water	Algae	Pseudokirchneriella subspicata	8.0-21.0	Nano silica suspension	24 ± 1 °C	72	Algal growth inhibition	IC50	34580	10	10	346	[5]
Water	Algae	Chlorella kessleri	5	Colloidal SAS in suspensions.	pH 7.7	96	Inhibition	IC50	8000000	1	10	800000	[6]
Water	Algae	Chlorella kessleri	26	Colloidal SAS in suspensions.	pH 7.8	96	Inhibition	IC50	71000000	1	10	7100000	[6]
Water	Algae	Chlorella kessleri	78	Colloidal SAS in suspensions.	pH 7.9	96	Inhibition	IC50	9100000	1	10	910000	[6]
Water	Algae	Chlorella autotrophica	20-50	nanoscale SiO2	25 °C	144	Growth	HONEC	1000000	1	1	1000000	[7]
Water	Algae	Scenedesmus obliquus	10.0-20.0	SiO2 particles with purity 99.5%	25 ± 1 °C.	96	Inhibition	EC20	216000	1	2	108000	[8]
Water	Invertebrate	Daphnia magna	10.0-20.0	Nano silica suspension	24-25 °C	24	Immobilization	EC50	148871	10	10	1489	[9]
Water	Invertebrate	Daphnia magna	10.0-20.0	Nano silica suspension	24-25 °C	24	Mortality	LC50	660943	10	10	6609	[9]
Water	Invertebrate	Daphnia magna	50 and 100	Amorphous plain silica nanoparticles and Fluorescently labeled silica NPs.	pH:6.04-6.28	48	Immobilization	HNOEC	> 1000000	10	1	> 100000	[4]
water	Invertebrate	Daphnia magna	10~20	SiO2 powder	24-25℃	24	immobilization	EC50	148871	10	10	1489	[10]
water	Invertebrate	Enchytraeus crypticus	24	SiO2	20℃	120	mortality	NOEC	>1000000	10	1	100000	[11]

Water	Invertebrate	Thamnocephalus platyurus	50 and 100	Amorphous plain silica nanoparticles and Fluorescently labeled silica NPs.	25 °C	24	Mortality	HNOEC	> 1000000	10	1	> 10000	[4]
Water	Unicellular	Tetrahymena termophila	8.0-21.0	Nano silica suspension	25 °C	24	Inhibition	HONEC	100000	10	1	10000	[5]
water	Unicellular	E. coli	14	SiO2 powder	23 ℃	14-20	growth inhibition	NOEC	100000	10	1	10000	[12]
water	Unicellular	B. subtilis	14	SiO2 powder	23 ℃	14-20	growth inhibition	NOEC	500000	10	1	50000	[12]
Water	Unicellular	Saccharomyces cerevisiae	10.0-20.0	Commercial nano silica particles with 99.5% purity	30 ℃	14	Cell damage	HONEC	1000000	10	1	100000	[13]
water	Vertebrate	Zebrafish	15, 30	SiO2	pH 6.5-7.5	120	mortality	HONEC	>213940	10	1	21394	[14]
water	Vertebrate	Zebrafish	70	SiO2	pH 6.5-7.5	120	mortality	LC50	178270	10	10	1783	[14]
water	Vertebrate	zebrafish larvae	15	SiO2 coated with 4% bovine serum albumin	28 °C	116	mortality	LC50	363400	10	10	3634	[15]
water	Vertebrate	zebrafish larvae	50	SiO2 coated with 4% bovine serum albumin	28 °C	116	mortality	LC50	312200	10	10	3122	[15]
water	Vertebrate	Zebrafish	15	SiO2 coated with 4% bovine serum albumin	pH7.0-7.2	168	mortality	LC50	782770	10	10	7828	[16]
water	Vertebrate	Zebrafish	50	SiO2 coated with 4% bovine serum albumin	pH7.0-7.2	168	mortality	LC50	637010	10	10	6370	[16]
Water	Vertebrate	Danio rerio, zebrafish	200	core-shell silica nanoparticles	26 °C	24	Survival and hatching	HONEC	200000	10	1	20000	[17]
Water	Vertebrate	Danio rerio, zebrafish	62	Silica nanoparticles prepared by Stöber method	28 °C	96	Mortality	LC20	200000	10	2	10000	[18]

Environmental compartment	**Taxonomic group**	***Test Organism***	**Particle Size (nm)**	**Test material**	**Test conditi-on**	**Exposure time (h)**	**Test endpoint**	**Ecoto-xic end-point**	**Concentration (μg/l)**	**Assessment Factor (AF) for the ex-trapolation from short to long term effect**	**Assessment Factor (AF) for the ex-trapolation from the observed effect into no effect con-centra-tions**	**Species sen-sitivity [μg/l]**	**Reference**
Water	Unicellular	*Escherichia coli (Migula)*	30-40	Fe2O3		2	Mortality	LC50	638300	10	10	6383	[19]
Water	Unicellular	*Escherichia coli*	8.0-21.0	Fe3O4 suspension	25 ºC	24	Inhibition	LOEC	100000	10	2	5000	[5]
Water	Unicellular	*Photobacterium phosphoreum*	7.57 ± 5.6	Fe3O4	pH 7	0.083-0.25	Bioluminescence re-duction	NOEC	> 52000	10	1	>5200	[20]
Water	Vertebrate	*Oryzias latipes*	27.2 ± 1.0	Fe3O4		168	Mortality	HNOEC	87200	10	1	8720	[21]
Water	Invertebrate	*Ceriodaphnia dubia*	21-890	Fe2O3	pH 7.8	48	Mortality	NOEC	200000	10	1	20000	[22]
Water	Vertebrate	*Danio rerio (emb-ryo)*	30	α-Fe2O3 NPs	26°C±1°C	168	Survival	NOEC	<50000	10	1	5000	[23]
Water	Vertebrate	*Danio rerio (egg)*	<50	Fe2O3	pH 8.3–8.6.	96	Mortality	NOEC	>1600000	10	1	160000	[24]
Water	Vertebrate	*Danio rerio (adult)*	<50	Fe2O3	pH 8.3–8.6.	96	Mortality	NOEC	>1600000	10	1	160000	[24]
Water	Invertebrate	*Daphnia magna*	6	Fe3O4 solution		48	Mortality	LC50	2300	10	10	23	[25]
Water	Invertebrate	*Daphnia magna*	5.7 ± 0.5	PVP-coated mag-netite (Fe3O4)		96	Immobilization	LOEC	100000	10	2	5000	[26]
Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Ascorbate-Fe3O4 coating	18-22	96	Immobilization	EC50	50700	10	10	507	[27]
Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Citrate-Fe3O4 coating	pH 7	96	Immobilization	EC50	68400	10	10	684	[27]
Water	Invertebrate	*Daphnia magna*	5.2 ± 0.4	Dextran-Fe3O4 coating	pH 7	96	Immobilization	EC50	27900	10	10	279	[27]
Water	Invertebrate	*Daphnia magna*	6.1 ± 0.6	Polymer-Fe3O4 coating	pH 7	96	Immobilization	NOEC	> 100000	10	1	10000	[27]

Water	Invertebrate	*Daphnia magna*	40-130	Fe2O3	20±2 °C	48	lethal effect	HONEC	20000	10	1	2000	[28]
Water	Invertebrate	*Brachionus plicatilis*	40-130	Fe2O3	20±2 °C	48	growth rate	HONEC	20000	10	1	2000	[28]
Water	Unicellular	*Pseudokirchneriella subcapita*	40-130	Fe2O3	20±2 °C	72	growth rate	HONEC	20000	1	1	20000	[28]
Water	Unicellular	*Pseudokirchneriella subcapita*	8.0-21.0	Fe3O4 suspension	24 ± 1 °C	72	Algal growth inhibition	EC50	1930	1	10	193	[5]
Water	Invertebrate	*Lemna minor*	40-130	Fe2O3	20±2 °C	7 days	growth rate	HONEC	20000	10	1	2000	[28]
Water	Unicellular	*Tetrahymena thermophila*	8.0-21.0	Fe3O4 suspension	25ºC	24	Inhibition	EC50	26030	10	10	260.3	[5]
Water	Unicellular	*Staphylococcus aureus*	8.0-21.0	Fe3O4 suspension	25ºC	24	Inhibition	LOEC	100000	10	2	5000	[5]
water	Unicellular	*Chlorella pyrenoidosa*	DLS 56 in water 64 in OECD medium	FeO	25℃	144	Algal growth inhibition	NOEC	>1000,000	1	1	1000000	[29]
water	Unicellular	*Chlorella pyrenonedosa*	30±5	α-Fe2O3	25℃	96	Algal growth inhibition	IC50	71000	1	10	7100	[30]
water	Unicellular	*Chlorella pyrenonedosa*	20±5	γ-Fe2O3	25℃	96	Algal growth inhibition	IC50	132000	1	10	13200	[30]
water	Unicellular	*Chlorella pyrenonedosa*	20±5	Fe3O4	25℃	96	Algal growth inhibition	IC50	33000	1	10	3300	[30]
water	Unicellular	*Stylonychia mytilus*	65	Fe3O4	20℃	24	Mortality	LOEC	9261	10	2	463	[31]

Environmental compartment	Taxonomic group	Test Organism	Particle Size (nm)	Test material	Test condition	Exposure time (h)	Test endpoint	Ecotoxic endpoint	Concentration (µg/l)	Assessment Factor (AF) for the extrapolation from short to long term effect	Assessment Factor (AF) for the extrapolation from the observed effect into no effect concentrations	Species sensitivity [µg/l]	Reference
water	unicellular	Staphylococcus aureu	BET 11.4	nAl2O3	25℃	24	growth	NOEC	100000	10	1	10000	[5]
water	unicellular	Escherichia coli	BET 11.4	nAl2O3	25℃	24	growth	NOEC	100000	10	1	10000	[5]
water	unicellular	E. coli	30-40	nAl2O3	NA	2	mortality	LC50	326100	10	10	3261	[19]
water	algae micro	Pseudokirchneriella subcapitata	BET 11.4	nAl2O3	24℃	72	growth inhibition	EC50	30800	10	10	308	[5]
water	algae	Pseudokirchnella subcapitata	50-150	nAl2O3,10% (w/w) dispersions in water.	20℃	4.5	efficiency of photosynthesis	NOEC	100000	10	1	10000	[32]
water?	unicellular	Saccharomyces cerevisiae	50	nAl2O3	pH=6.5, 30℃	10~14	inhibition	HONEC	1000000	10	1	100000	[13]
water	Invertebrate	C. dubia.	5	nAl2O3,γ-phase	pH=7.8	48	mortality	EC15	150000	10	2	7500	[22]
water	invertebrate	C. dubia	ad:<50; TEM 40-100	nAl2O3, γ-phase	NA	72	mortality	LC50	74300	10	10	743	[33]
water	unicellular	Stylonychia mytilus	NA	nAl2O3,α-phase	20 ±2℃	24	mortality	LOEC	1275000	10	2	63750	[31]
water	unicellular	Paramecium multimicronucleatum	ad: 30-40; TEM 83.5	nAl2O3,α-phase	NA	48	mortality	LC50	9269200	1	10	926920	[34]
water	algae	Scenedesmus sp.	<50	nAl2O3	27℃	72	growth inhibition	NOEC	2100	10	1	210	[35]
water	algae	Chlorella sp.	<50	nAl2O3	27℃	72	growth inhibition	NOEC	1800	10	1	180	[35]
water	invertebrate	Chydorus sphaericus	50-150	nAl2O3,10% (w/w) dispersions in water.	20℃	48	mortality	NOEC	100000	10	1	10000	[32]

water	invertebrate	C. dubia	NA	nAl2O3,γ-phase	pH=7.8, 25±1℃	48	mortality	LC10	100000	10	2	5000	[36]
water	unicellular	Tetrahymena thermophila	BET 11.4	nAl2O3	25℃	24	inhibition	NOEC	100000	10	1	10000	[5]
water	invertebrate	C. elegans	60	nAl2O3	pH=5.6, 20±1℃	24	mortality	LC50	81600	10	10	816	[37]
water	invertebrate	Caenorhabditis elegans	60	nAl2O3	NA	24	mortality	LC50	81600	10	10	816	[37]
water	invertebrate	Caenorhabditis elegans (young adult)	60	nAl2O3	pH=5.6	24	mortality	LC10	6300	10	2	315	[38]
water	invertebrate	Caenorhabditis elegans (L4-Larval stage)	60	nAl2O3	pH=5.6	24	mortality	LC10	6300	10	2	315	[38]
water	invertebrate	Caenorhabditis elegans (L1-Larval stage)	60	nAl2O3	pH=5.6	24	mortality	LC50	51000	10	10	510	[38]
water	invertebrate	D. magna	80	nAl2O3	23 ± 2 ℃	48	immobilization	EC50	114357	10	10	1143.57	[39]
water	vertebrate	Danio rerio	80	nAl2O3	26±1ºC	96	survival	HONEC	1000000	10	1	100000	[39]
water	vertebrate	Danio rerio (adult)	<50	nAl2O3	pH=8.3-8.6,25± 1 ℃	96	mortality	NOEC	800000	10	1	80000	[24]
water	vertebrate	Danio rerio (eggs)	<50	nAl2O3	pH=8.3-8.6,25± 1 ℃	96	mortality	NOEC	800000	10	1	80000	[24]

Environmental compartment	**Taxonomic group**	**Test Organism**	**Particle Size (nm)**	**Test material**	**Test condition**	**Exposure time (h)**	**Test endpoint**	**Ecotoxic endpoint**	**Concentration (µg/l)**	**Assessment Factor (AF) for the extrapolation from short to long term effect**	**Assessment Factor (AF) for the extrapolation from the observed effect into no effect concen-**	**Species sensitivity [µg/l]**	**Reference**

											tra-tions		
Water	Algae	Pseudokirchneriella subcapitata	10~20	CeO2 NP	cool-white light,synthetic fresh water,pH 6.46-6.3	72	growth inhibition	IC50	7900	10	10	79	[40]
Water	Algae	Pseudokirchneriella subcapitata	10~20	CeO2 NP	cool-white light,synthetic fresh water,pH 6.46-6.3	72	growth inhibition	IC50	7600	10	10	76	[40]
Water	Algae	Pseudokirchneriella subcapitata	10~20	CeO2 NP	cool-white light,synthetic fresh water + 40mg/L DOC,pH 6.46-6.3	72	growth inhibition	IC50	100000	10	10	1000	[40]
Water	Algae	Pseudokirchneriella subcapitata	34.00	CeO2 NP	cool-white light,synthetic fresh water,pH 6.46-6.3	72	growth inhibition	IC50	9500	10	10	95	[40]
Water	Algae	Pseudokirchneriella subcapitata	34.00	CeO2 NP	cool-white light, synthetic fresh water + 40mg/L DOC,pH 6.46-6.3	72	growth inhibition	IC50	100000	10	10	1000	[40]
Water	Algae	Pseudokirchneriella subcapitata	12.00	CeO2 NP	cool-white light,synthetic fresh water,pH 6.46-6.3	72	growth inhibition	IC50	28000	10	10	280	[40]
Water	Algae	Pseudokirchneriella subcapitata	12.00	CeO2 NP	cool-white light,synthetic fresh water + 40mg/L DOC,pH 6.46-6.3	72	growth inhibition	IC50	100000	10	10	1000	[40]
Water	Algae	Pseudokirchneriella subcapitata	4.0-10.0	poly (acrylic acid)-stabilised colloidal dispersion of CeO2 particles	21-24 °C	72	Biomass production	EC10	5.3	10	2	0.265	[41]
Water	Algae	Pseudokirchneriella subcapitata	10	Non aged nano-CeO2 suspension,coated with triammonium citrate layer, 10 nm	22 ± 1 °C	72	Growth Inhibition	EC10	700	10	2	35	[42]
Water	Algae	Pseudokirchneriella subcapitata	10	3 days aged nano-CeO2 suspension,coated with triammonium citrate layer, 10 nm	22 ± 1 °C	72	Growth Inhibition	EC10	500	10	2	25	[42]
Water	Algae	Pseudokirchneriella subcapitata	10	30 days aged nano-CeO2 suspension,coated with triammonium citrate layer, 10 nm	22 ± 1 °C	72	Growth Inhibition	EC10	1800	10	2	90	[42]

Water	Algae	Pseudokirchneriella subcapitata	12	Nano-CeO2 powder, uncoated	22°C	96	Growth inhibition, OD at 640 nm	EC50	12800	1	10	1280	[43]
Water	Algae	Pseudokirchneriella subcapitata	13	Nano-CeO2 powder, uncoated	22°C	96	Growth inhibition, OD at 640 nm	EC50	950	1	10	95	[43]
Water	Algae	Pseudokirchneriella subcapitata	22	Nano-CeO2 powder, uncoated	22°C	96	Growth inhibition, OD at 640 nm	EC50	880	1	10	88	[43]
Water	Algae	Pseudokirchneriella subcapitata	28	Nano-CeO2 powder, uncoated	22°C	96	Growth inhibition, OD at 640 nm	EC50	8960	1	10	896	[43]
Water	Algae	Pseudokirchneriella subcapitata	7–25	uncoated CeO2 NP	24°C, pH = 6.5±0.1	72	Growth inhibition	IC50	10300	10	10	103	[44]
Water	Algae	Pseudokirchneriella subcapitata	14	CeO2 NP	25 °C	72	Growth inhibition	NOEC	3200	10	1	320	[45]
Water	Algae	Pseudokirchneriella subcapitata	20	CeO2 NP	25 °C	72	Growth inhibition	NOEC	3200	10	1	320	[45]
Water	Algae	Pseudokirchneriella subcapitata	29	CeO2 NP	25 °C	72	Growth inhibition	NOEC	3200	10	1	320	[45]
Water	Algae	Pseudokirchneriella subcapitata	14	CeO2 NP	standard OECD algal medium, pH 7.4, 25 °C	48	Growth inhibition	EC20	3500	10	2	175	[46]
Water	Algae	Pseudokirchneriella subcapitata	14	CeO2 NP	standard OECD algal medium with 3.6 mM 3-(N-Morpholino)propanesulfonic acid (MOPS) buffer and 0 mg C/L Bihain NOM, pH 7.47, , 25 °C	48	Growth inhibition	EC20	4700	10	2	235	[46]
Water	Algae	Pseudokirchneriella subcapitata	14	CeO2 NP	standard OECD algal medium with 3.6 mM 2-(Cyclohexylamino)ethanesulfonic acid (CHES) buffer and 5.43 mg C/L Bihain NOM, pH 6,25 °C	48	Growth inhibition	EC20	395800	10	2	19790	[46]
Water	Algae	Pseudokirchneriella subcapitata	14	CeO2 NP	standard OECD algal medium with 3.6 mM 2-(Cyclohexylamino)ethanesulfonic acid (CHES) buffer and 5.43 mg C/L Bihain NOM, pH 7.47,25 °C	48	Growth inhibition	EC20	395800	10	2	19790	[46]
Water	Algae	Pseudokirchneriella subcapitata	50-150	CeO2 suspension	20 °C	4.5	Efficiency of photosynthesis	NOEC	100000	10	1	10000	[32]

Water	Algae	Pseudokirchneriella subcapitat	33~49	NP CeO2	21-24°C,OECD algae growth medium	72	Growht inhibition	EC50	1240	10	10	12.4	[47]
Water	Algae	Pseudokirchneriella subcapitat	33~49	NP CeO2	21-24°C,OECD algae growth medium + 8 mg/L SR-NOM	72	Growht inhibition	EC10	16200	10	2	810	[47]
Water	Algae	Pseudokirchneriella subcapitat	33~49	NP CeO2	21-24°C,OECD algae growth medium + 20 mg/L SR-NOM	72	Growht inhibition	EC10	38000	10	2	1900	[47]
Water	Algae	Raphidocelis subcapitata	38	CeO2 NP	24 ± 1 °C	72	growth inhibition	EC50	8200	10	10	82	[48]
Water	Invertebrate	Daphnia pulex	3±1	CeO2 NP suspension	20 ± 2 °C,pH=8.5	96	Immobility	EC50	780	10	10	7.8	[49]
Water	Invertebrate	Daphnia similis	3±1	CeO2 NP suspension	20 ± 2 °C,pH=8.5	48	Immobility	EC50	260	10	10	2.6	[49]
Water	Algae	Nitzschia palea	2.0-5.0	suspension of CeO2 NPs coated with a tri-ammonium citrate layer.	20 ± 1 °C, pH 6.4-6.6	48	Growth	NOEC	100000	10	1	10000	[50]
Water	Algae	Nitzschia palea	20.0-60.0	CeO2 NP	20 ± 1 °C, , pH 6.4-6.6	48	Growth	NOEC	100000	10	1	10000	[50]
Water	Vertebrate	Xenopus laevis larvae	2.0-5.0	suspension of CeO2 NPs coated with a tri-ammonium citrate layer.	22 ± 2 °C	288	Mortality	NOEC	10000	10	1	1000	[50]
Water	Vertebrate	Xenopus laevis larvae	20.0-60.0	CeO2 NP	22 ± 2 °C	288	Mortality	LC35	10000	10	10	100	[50]
Water	Vertebrate	Pleurodeles waltl	2.0-5.0	suspension of CeO2 NPs coated with a tri-ammonium citrate layer.	22 ± 2 °C	288	Mortality	NOEC	10000	10	1	1000	[50]
Water	Vertebrate	Pleurodeles waltl	20.0-60.0	CeO2 NP	22 ± 2 °C	288	Mortality	NOEC	10000	10	1	1000	[50]
Water	Invertebrate	Chironomus riparius	20.0-60.0	CeO2 NP	21 ± 1 °C	168	Growth inhibition	NOEC	100000	10	1	10000	[50]
Water	Invertebrate	Chironomus riparius	2.0-5.0	suspension of CeO2 NPs coated with a tri-ammonium	21 ± 1 °C	168	Growth inhibition	NOEC	100000	10	1	10000	[50]

				citrate layer.									
Water	Invertebrate	Chironomus riparius	15	Nano-CeO2	20 ± 1 °C	24	Mortality	EC15	1000	10	2	50	[51]
Water	Invertebrate	Chironomus riparius	30	Nano-CeO2	20 ± 1 °C	24	Mortality	EC10	1000	10	2	50	[51]
Water	Invertebrate	Ceriodaphnia dubia	10.0-30.0	CeO2 NP	25 °C, pH 6.8	24	Mortality	NOEC	250000	10	1	25000	[52]
Water	Invertebrate	Ceriodaphnia dubia	10.0-30.0	CeO2 NP	25 °C, pH 7.8	24	Mortality	NOEC	1000000	10	1	100000	[52]
Water	Algae	Desmodesmus subspicatus	58	CeO2 NP	23 ± 2 °C	72	Growht inhibition	EC50	490000	10	10	4900	[53]
Water	Algae	Anabaena sp.	12	Nano-CeO2 powder, uncoated	Pure water medium, pH 6, 28 °C	24	Luminescence inhibition	EC50	6300	10	10	63	[43]
Water	Algae	Anabaena sp.	13	Nano-CeO2 powder, uncoated	Pure water medium, pH 6, 28 °C	24	Luminescence inhibition	EC50	560	10	10	5.6	[43]
Water	Algae	Anabaena sp.	22	Nano-CeO2 powder, uncoated	Pure water medium, pH 6, 28 °C	24	Luminescence inhibition	EC50	270	10	10	2.7	[43]
Water	Algae	Anabaena sp.	28	Nano-CeO2 powder, uncoated	Pure water medium, pH 6, 28 °C	24	Luminescence inhibition	EC50	7500	10	10	75	[43]
Water	Algae	Anabaena sp.	12	Nano-CeO2 powder, uncoated	cyanobacterial culture medium AA/8+ N 1/10 buffered with 2mM HEPES, pH=6, 28°C	24	Luminescence inhibition	EC10	62	10	2	3.1	[43]
Water	Algae	Anabaena sp.	13	Nano-CeO2 powder, uncoated	cyanobacterial culture medium AA/8+ N 1/10 buffered with 2mM HEPES, pH 6, 28C	24	Luminescence inhibition	EC10	8	10	2	0.4	[43]
Water	Algae	Anabaena sp.	22	Nano-CeO2 powder, uncoated	cyanobacterial culture medium AA/8+ N 1/10 buffered with 2mM HEPES, pH 6, 28°C	24	Luminescence inhibition	EC10	57	10	2	2.85	[43]
Water	Algae	Anabaena sp.	28	Nano-CeO2 powder, uncoated	cyanobacterial culture medium AA/8+ N 1/10 buffered with 2mM HEPES, pH 6, 28°C	24	Luminescence inhibition	EC10	89	10	2	4.45	[43]

Water	Invertebrate	Caenorhabditis elegans	15	nano-CeO2 suspension	20 °C	24	Mortality	LC20	1000	10	2	50	[54]
Water	Invertebrate	Caenorhabditis elegans	45	nano-CeO2 suspension	20 °C	24	Mortality	NOEC	1000	10	1	100	[54]
Water	Invertebrate	Caenorhabditis elegans	hydrodynamic 53.34 ± 3.12	CeO2 NP	20 °C,pH=7.17-7.19	72	Growth inhibition	LOEC	2500	10	2	125	[55]
Water	Invertebrate	Ceriodaphnia affinis	10-100	Nano-CeO2	21 ± 3 °C, pH 7.5–8.0	168	Mortality	LC50	97100	10	10	971	[56]
Water	unicelluar	nanoflagellates Bodo saltans Ehrenberg	10-100	Nano-CeO2	21 ± 3 °C, pH 7.5–8.0	96	population	LOEC	20000	1	1	20000	[56]
Water	Invertebrate	Thamnocephalus platyurus	14	CeO2 NP	25 °C	24	Mortality	NOEC	5000000	10	1	500000	[45]
Water	Invertebrate	Thamnocephalus platyurus	20	CeO2 NP	25 °C	24	Mortality	NOEC	5000000	10	1	500000	[45]
Water	Invertebrate	Thamnocephalus platyurus	29	CeO2 NP	25 °C	24	Mortality	NOEC	5000000	10	1	500000	[45]
Water	Invertebrate	Daphnia magna	14	CeO2 NP	20±1°C	24	Survival	NOEC	32000	10	1	3200	[45]
Water	Invertebrate	Daphnia magna	20	CeO2 NP	20±1°C	24	Survival	NOEC	32000	10	1	3200	[45]
Water	Invertebrate	Daphnia magna	29	CeO2 NP	20±1°C	24	Survival	NOEC	56000	10	1	5600	[45]
Water	Invertebrate	Daphnia magna	<25	nano-CeO2 suspension	NA	504	Mortality	NOEC	3000	10	1	300	[57]
Water	Invertebrate	Daphnia magna	6.5	CeO2 NP	pH 7	48	Immobility	LC50	12000	10	10	120	[57]
Water	Invertebrate	Daphnia magna	15	Nano-CeO2	18-22 °C	96	Mortality	EC10	1000	10	2	50	[51]

Water	Invertebrate	Daphnia magna	30	Nano-CeO2	18-22 °C	96	Mortality	NOEC	1000	10	1	100	[51]
Water	Vertebrate	Danio rerio	<10	poly(acrylic acid) (PAA)-coated **CeO2**	28 ± 1 °C, pH=7.5-7.6	72	Mortality	NOEC	2000000	10	1	200000	[58]
Water	Vertebrate	Danio rerio	14	CeO2 NP	28 °C	24	Mortality	NOEC	200000	10	1	20000	[45]
Water	Vertebrate	Danio rerio	20	CeO2 NP	28 °C	24	Mortality	NOEC	200000	10	1	20000	[45]
Water	Vertebrate	Danio rerio	29	CeO2 NP	28 °C	24	Mortality	NOEC	200000	10	1	20000	[45]
Water	Invertebrate	Chydorus sphaericus	50-150	CeO2 suspension	20 °C	48	Mortality	NOEC	100000	10	1	10000	[32]
Water													

Environmental compartment	**Taxonomic group**	**Test Organism**	**Particle Size (nm)**	**Test material**	**Test condition**	**Exposure time (h)**	**Test endpoint**	**Ecotoxic endpoint**	**Concentration (µg/l)**	**Concentration (µg QD/l)**	**Assessment Factor (AF) for the extrapolation from short to long term effect**	**Assessment Factor (AF) for the extrapolation from the observed effect into no effect concentrations**	**Species sensitivity [µg/l]**	**Reference**
Water	Invertebrate	Ceriodaphnia dubia	10.0-25.0	Carboxyl Quantum Dots	25±1℃	48	Mortality	NOEC	110	110	10	1	11	[59]
Water	Algae	Pseudokirchneriella subcapitata	10.0-25.0	Carboxyl Quantum Dots	25±1℃	96	Mortality	LC50	37.1	37.1	1	10	3.71	[59]
Water	Invertebrate	Caenorhabditis elegans(L4)	3.4±0.72	CdSe core QD	20℃	72	Brood size	EC26	20000 Cd	34048.57219	10	10	340.4857219	[60]
Water	Invertebrate	Elliptio complanata	5.0-10.0	CdTe QDs	15℃	24	Mortality	NOEC	8000	8000	10	1	800	[61]

Water	Invertebrate	Daphnia magna	hydrodynamic size 65.9	GA-QD	20±1℃	48	Mortality	EC50	95.9	95.9	10	10	0.959	[62]
Water	Invertebrate	Daphnia magna	hydrodynamic size 65.9	GA-QD	20±1℃, UVB	48	Mortality	EC50	58.5	58.5	10	10	0.585	[62]
Water	Invertebrate	Daphnia magna	NA	MPA-QD	20±1℃	48	Mortality	NOEC	94.9	94.9	10	1	9.49	[62]
Water	Invertebrate	Daphnia magna	NA	MPA-QD	20±1℃, UVB	48	Mortality	NOEC	94.9	94.9	10	1	9.49	[62]
Water	Invertebrate	Daphnia magna	5	Red polyethylene oxide (PEO)-coated CdSe/ZnS QDs	21±2℃, pH=7.9±0.8	48	Mortality	EC50	3840	3840	10	10	38.4	[63]
Water	Invertebrate	Daphnia magna	2	Green polyethylene oxide coated CdSe/ZnS QDs	21±2℃, pH=7.9±0.8	48	Mortality	EC50	770	770	10	10	7.7	[63]
Water	Invertebrate	Daphnia magna	2	Green mercaptoundecanoic acid (MUA)-coated CdSe/ZnS QDs	21±2℃, pH=7.9±0.8	48	Mortality	EC50	350	350	10	10	3.5	[63]
Water	Invertebrate	Daphnia magna	5	Red 11-mercaptoundecanoic acid (MUA)-coated CdSe/ZnS QDs	21±2℃, pH=7.9±0.8	48	Mortality	EC50	110	110	10	10	1.1	[63]
Water	Algae	C. reinhardtii	3.5~4.5	CdTe quantum dots	25 ± 0.5℃	72	growth	EC50	5000	5000	10	10	50	[64]
Water	Vertebrate	Danio rerio	2~5	graphene quantum dots	28±1℃	116	Mortality	LOEC	50	50	10	2	2.5	[64]
Water	Vertebrate	Danio rerio	3.5	TGA-CdTe QDs	28±0.5℃	120	mortality	EC8	6000	6000	10	2	300	[65]
Water	Vertebrate	Danio rerio	4	CdTe QDs	28℃	96	Mortality	NOEC	2.5 nM	1.502	10	1	0.1502	[18]

Reference

[1] Van Hoecke K, De Schamphelaere KAC, Van der Meeren P, Lucas S, Janssen CR. 2008. Ecotoxicity of silica nanoparticles to the green alga Pseudokirchneriella subcapitata: Importance of surface area. *Environ Toxicol Chem* 27:1948-1957.

[2] Van Hoecke K, De Schamphelaere KA, Ramirez-Garcia S, Van der Meeren P, Smagghe G, Janssen CR. 2011. Influence of alumina coating on characteristics and effects of SiO2 nanoparticles in algal growth inhibition assays at various pH and organic matter contents. *Environ Int* 37:1118-1125.

[3] Metzler DM, Erdem A, Tseng YH, Huang CP. 2012. Responses of Algal Cells to Engineered Nanoparticles Measured as Algal Cell Population, Chlorophyll a, and Lipid Peroxidation: Effect of Particle Size and Type. *Journal of Nanotechnology* 2012:12.

[4] Casado MP, Macken A, Byrne HJ. 2013. Ecotoxicological assessment of silica and polystyrene nanoparticles assessed by a multitrophic test battery. *Environ Int* 51:97-105.

[5] Aruoja V, Pokhrel S, Sihtmae M, Mortimer M, Madler L, Kahru A. 2015. Toxicity of 12 metal-based nanoparticles to algae, bacteria and protozoa. *Environ Sci-Nano* 2:630-644.

[6] Fujiwara K, Suematsu H, Kiyomiya E, Aoki M, Sato M, Moritoki N. 2008. Size-dependent toxicity of silica nano-particles to Chlorella kessleri. *Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering* 43:1167-1173.

[7] Ji J, Long ZF, Lin DH. 2011. Toxicity of oxide nanoparticles to the green algae Chlorella sp. *Chem Eng J* 170:525-530.

[8] Wei CX, Zhang YB, Guo J, Han B, Yang X, Yuan JL. 2010. Effects of silica nanoparticles on growth and photosynthetic pigment contents of Scenedesmus obliquus. *Environmental Science* 22:6.

[9] Yang SY, Ye RF, Han B, Wei CX, Yang X. 2014. Ecotoxicological Effect of Nano-silicon Dioxide Particles on Daphnia Magna. *Integr Ferroelectr* 154:64-72.

[10] Han B, Wei C, Yang D, Yu X, Yang X. 2009. Acute Toxicity of Suspension of Nanosized Silicon Dioxide Particles to Daphnia Magna. *Bioinformatics and Biomedical Engineering*. IEEE, Beijing China.

[11] Bicho RC, Soares AM, Nogueira HIS, Amorim MJ. 2016. Effects of europium polyoxometalate encapsulated in silica nanoparticles (nanocarriers) in soil invertebrates. *J Nanopart Res* 18:7.

[12] Adams LK, Lyon DY, Alvarez PJJ. 2006. Comparative eco-toxicity of nanoscale TiO_2, SiO_2, and ZnO water suspensions. *Water Res* 40:3527-3532.

[13] Garcia-Saucedo C, Field JA, Otero-Gonzalez L, Sierra-Alvarez R. 2011. Low toxicity of HfO2, SiO2, Al2O3 and CeO2 nanoparticles to the yeast, Saccharomyces cerevisiae. *J Hazard Mater* 192:1572-1579.

[14] Lacave JM, Retuerto A, Vicario-Pares U, Gilliland D, Oron M, Cajaraville MP, Orbea A. 2016. Effects of metal-bearing nanoparticles (Ag, Au, CdS, ZnO, SiO2) on developing zebrafish embryos. *Nanotechnology* 27:325102.

[15] Wang Y, Li X, Zhang Y, Feng X. 2014. Developmental and cartilaginous effects of protein-coated SiO2 nanoparticle corona complexes on zebrafish larvae. *Royal Society of Chemistry* 4:8.

[16] Xue JY, Li X, Sun MZ, Wang YP, Wu M, Zhang CY, Wang YN, Liu B, Zhang YS, Zhao X, Feng XZ. 2013. An assessment of the impact of SiO2 nanoparticles of different sizes on the rest/wake behavior and the developmental profile of zebrafish larvae. *Small* 9:3161-3168.

[17] Fent K, Weisbrod CJ, Wirth-Heller A, Pieles U. 2010. Assessment of uptake and toxicity of fluorescent silica nanoparticles in zebrafish (Danio rerio) early life stages. *Aquat Toxicol* 100:218-228.

[18] Duan JC, Yu YB, Shi HQ, Tian LW, Guo CX, Huang PL, Zhou XQ, Peng SQ, Sun ZW. 2013. Toxic Effects of Silica Nanoparticles on Zebrafish Embryos and Larvae. *Plos One* 8.

[19] Hu X, Cook S, Wang P, Hwang HM. 2009. In vitro evaluation of cytotoxicity of engineered metal oxide nanoparticles. *The Science of the total environment* 407:3070-3072.

[20] Barrena R, Casals E, Colon J, Font X, Sanchez A, Puntes V. 2009. Evaluation of the ecotoxicity of model nanoparticles. *Chemosphere* 75:850-857.

[21] Chen PJ, Tan SW, Wu WL. 2012. Stabilization or Oxidation of Nanoscale Zerovalent Iron at Environmentally Relevant Exposure Changes Bioavailability and Toxicity in Medaka Fish. *Environmental Science & Technology* 46:8431-8439.

[22] Hu J, Wang D, Forthaus BE, Wang J. 2012. Quantifying the effect of nanoparticles on As(V) ecotoxicity exemplified by nano-Fe2 O3 (magnetic) and nano-Al2 O3. *Environ Toxicol Chem* 31:2870-2876.

[23] Zhu XS, Tian SY, Cai ZH. 2012. Toxicity Assessment of Iron Oxide Nanoparticles in Zebrafish (Danio rerio) Early Life Stages. *Plos One* 7:e46286.

[24] Kovriznych JA, Sotnikova R, Zeljenkova D, Rollerova E, Szabova E, Wimmerova S. 2013. Acute toxicity of 31 different nanoparticles to zebrafish (Danio rerio) tested in adulthood and in early life stages - comparative study. *Interdisciplinary toxicology* 6:67-73.

[25] Garcia A, Espinosa R, Delgado L, Casals E, Gonzalez E, Puntes V, Barata C, Font X, Sanchez A. 2011. Acute toxicity of cerium oxide, titanium oxide and iron oxide nanoparticles using standardized tests. *Desalination* 269:136-141.

[26] Filser J, Arndt D, Baumann J, Geppert M, Hackmann S, Luther EM, Pade C, Prenzel K, Wigger H, Arning J, Hohnholt MC, Koser J, Kuck A, Lesnikov E, Neumann J, Schutrumpf S, Warrelmann J, Baumer M, Dringen R, von Gleich A, Swiderek P, Thoming J. 2013. Intrinsically green iron oxide nanoparticles? From synthesis via (eco-)toxicology to scenario modelling. *Nanoscale* 5:1034-1046.

[27] Baumann J, Koser J, Arndt D, Filser J. 2014. The coating makes the difference: acute effects of iron oxide nanoparticles on Daphnia magna. *The Science of the total environment* 484:176-184.

[28] Nogueira V, Lopes I, Rocha-Santos TAP, Rasteiro MG, Abrantes N, Goncalves F, Soares AMVM, Duarte AC, Pereira R. 2015. Assessing the ecotoxicity of metal nano-oxides with potential for wastewater treatment. *Environ Sci Pollut R* 22:13212-13224.

[29] Karunakaran G, Gusev A, Kolesnikov E, Kuznetsov D, Jagathambal M. 2017. Assessment of FeO and MnO nanoparticles toxicity on Chlorella pyrenoidosa. *J Nanosci Nanotechno* 17:9.

[30] Lei C, Zhang L, Yang K, Zhu L, Lin D. 2016. Toxicity of iron-based nanoparticles to green algae: Effects of particle size, crystal phase, oxidation state and environmental aging. *Environmental pollution* 218:505-512.

[31] Kosyan DB, Rusakova EA, Sizova EA, Yausheva EV, Miroshnikov SA. 2016. TOXIC EFFECT AND MECHANISMS OF NANOPARTICLES ON FRESHWATER INFUSORIA. *International Journal of GEOMATE* 11:7.

[32] Velzeboer I, Hendriks AJ, Ragas AMJ, Van de Meent D. 2008. Aquatic ecotoxicity tests of some nanomaterials. *Environ Toxicol Chem* 27:1942-1947.

[33] Pakrashi S, Dalai S, Humayun A, Chakravarty S, Chandrasekaran N. 2013. Amitava MukherjeeCeriodaphnia dubia as a potential bio-indicator for assessing acute aluminum oxide nanoparticle toxicity in fresh water environment. *PLoS ONE* 8.

[34] Li K, Chen Y, Zhang W, Pu Z, Jiang L, Chen Y. 2012. Surface interactions affect the toxicity of engineered metal oxide nanoparticles toward Paramecium. *Chem Res Toxicol* 25:1675-1681.

[35] Sadiq IM, Pakrashi S, Chandrasekaran N, Mukherjee A. 2011. Studies on toxicity of aluminum oxide (Al2O3) nanoparticles to microalgae species: Scenedesmus sp and Chlorella sp. *J Nanopart Res* 13:3287-3299.

[36] Hu J, Wang DM, Forthaus BE, Wang JM. 2012. Quantifying the effect of nanoparticles on As(V) ecotoxicity exemplified by nano-Fe2O3 (magnetic) and nano-Al2O3. *Environ Toxicol Chem* 31:2870-2876.

[37] Wang H, Wick RL, Xing B. 2009. Toxicity of nanoparticulate and bulk ZnO, Al_2O_3 and TiO_2 to the nematode Caenorhabditis elegans. *Environ Pollut* 157:1171-1177.

[38] Wu S, Lu J, Rui Q, Yu S, Cai T, Wang D. 2011. Aluminum nanoparticle exposure in L1 larvae results in more severe lethality toxicity than in L4 larvae or young adults by strengthening the formation of stress response and intestinal lipofuscin accumulation in nematodes. *Environmental toxicology and pharmacology* 31:10.

[39] Zhu XS, Zhu L, Chen YS, Tian SY. 2009. Acute toxicities of six manufactured nanomaterial suspensions to Daphnia magna. *J Nanopart Res* 11:67-75.

[40] Angel BM, Vallotton P, Apte SC. 2015. On the mechanism of nanoparticulate CeO2 toxicity to freshwater algae. *Aquat Toxicol* 168:90-97.

[41] Booth A, Storseth T, Altin D, Fornara A, Ahniyaz A, Jungnickel H, Laux P, Luch A, Sorensen L. 2015. Freshwater dispersion stability of PAA-stabilised cerium oxide nanoparticles and toxicity towards Pseudokirchneriella subcapitata. *The Science of the total environment* 505:596-605.

[42] Manier N, Bado-Nilles A, Delalain P, Aguerre-Chariol O, Pandard P. 2013. Ecotoxicity of non-aged and aged CeO2 nanomaterials towards freshwater microalgae. *Environmental pollution* 180:63-70.

[43] Rodea-Palomares I, Boltes K, Fernandez-Pinas F, Leganes F, Garcia-Calvo E, Santiago J, Rosal R. 2011. Physicochemical Characterization and Ecotoxicological Assessment of CeO2 Nanoparticles Using Two Aquatic microorganisms. *Toxicol Sci* 119:135-145.

[44] Rogers NJ, Franklin NM, Apte SC, Batley GE, Angel BM, Lead JR, Baalousha M. 2010. Physico-chemical behaviour and algal toxicity of nanoparticulate CeO2 in freshwater. *Environmental Chemistry* 7:50-60.

[45] Van Hoecke K, Quik JTK, Mankiewicz-Boczek J, De Schamphelaere KAC, Elsaesser A, Van der Meeren P, Barnes C, McKerr G, Howard CV, Van De Meent D, Rydzynski K, Dawson KA, Salvati A, Lesniak

A, Lynch I, Silversmit G, De Samber B, Vincze L, Janssen CR. 2009. Fate and Effects of CeO2 Nanoparticles in Aquatic Ecotoxicity Tests. *Environmental Science & Technology* 43:4537-4546.

[46] Van Hoecke K, De Schamphelaere KAC, Van der Meeren P, Smagghe G, Janssen CR. 2011. Aggregation and ecotoxicity of CeO_2 nanoparticles in synthetic and natural waters with variable pH, organic matter concentration and ionic strength. *Environmental pollution* 159:970-976.

[47] Cerrillo C, Barandika G, Igartua A, Areitioaurtena O, Mendoza G. 2016. Towards the standardization of nanoecotoxicity testing: Natural organic matter 'camouflages' the adverse effects of TiO2 and CeO2 nanoparticles on green microalgae. *The Science of the total environment* 543:95-104.

[48] Joonas E, Aruoja V, Olli K, Syvertsen-Wiig G, Vija H, Kahru A. 2017. Potency of (doped) rare earth oxide particles and their constituent metals to inhibit algal growth and induce direct toxic effects. *Science of the Total Environment* 593-594:9.

[49] Artells E, Issartel J, Auffan M, Borschneck D, Thill A, Tella M, Brousset L, Rose J, Bottero JY, Thiery A. 2013. Exposure to cerium dioxide nanoparticles differently affect swimming performance and survival in two daphnid species. *Plos One* 8:e71260.

[50] Bour A, Mouchet F, Verneuil L, Evariste L, Silvestre J, Pinelli E, Gauthier L. 2015. Toxicity of CeO2 nanoparticles at different trophic levels--effects on diatoms, chironomids and amphibians. *Chemosphere* 120:7.

[51] Lee SW, ManKim S, Choi J. 2009. Genotoxicity and ecotoxicity assays using the freshwater crustacean Daphnia magna and the larva of the aquatic midge Chironomus riparius to screen the ecological risks of nanoparticle exposure. *Environmental toxicology and pharmacology* 28:6.

[52] Hu J, Wang DM, Wang JT, Wang JM. 2012. Toxicity of lead on Ceriodaphnia dubia in the presence of nano-CeO2 and nano-TiO2. *Chemosphere* 89:6.

[53] Ľubomíra KUZNÍKOVÁ KD, Lubomír PAVELEK2, Jana KUKUTSCHOVÁ1. 2014. SYNTHESIS, CHARACTERIZATION AND ACUTE AQUATIC TOXICITY OF CERIUM OXIDE NANOPARTICLES TO FRESHWATER GREEN ALGAE. *NANOCON*. Brno, Czech Republic, p. 6.

[54] Roh JY, Park YK, Park K, Choi J. 2010. Ecotoxicological investigation of CeO2 and TiO2 nanoparticles on the soil nematode Caenorhabditis elegans using gene expression, growth, fertility, and survival as endpoints. 29:6.

[55] Arnold MC, Badireddy AR, Wiesner MR, Di Giulio RT, Meyer JN. 2013. Cerium oxide nanoparticles are more toxic than equimolar bulk cerium oxide in Caenorhabditis elegans. *Archives of environmental contamination and toxicology* 65:10.

[56] Tomilina II, Gremyachikh VA, Myl'nikov AP, Komov VT. 2011. Changes in biological characteristics of freshwater heterotrophic flagellates and cladocerans under the effect of metal oxide nano- and microparticles. *Inland Water Biology* 4:9.

[57] Gaiser BK, Biswas A, Rosenkranz P, Jepson MA, Lead JR, Stone V, Tyler CR, Fernandes TF. 2011. Effects of silver and cerium dioxide micro- and nano-sized particles on Daphnia magna. *Journal of environmental monitoring : JEM* 13:1227-1235.

[58] Felix LC, Ortega VA, Ede JD. 2013. Physicochemical Characteristics of Polymer-Coated Metal-Oxide Nanoparticles and their Toxicological Effects on Zebrafish (Danio rerio) Development. 47:8.

[59] Bouldin JL, Ingle TM, Sengupta A, Alexander R, Hannigan RE, Buchanan RA. 2008. Aqueous toxicity and food chain transfer of quantum Dots (TM) in freshwater algae and Ceriodaphnia dubia. *Environ Toxicol Chem* 27:1958-1963.

[60] Contreras EQ, Cho MJ, Zhu HG, Puppala HL, Escalera G, Zhong WW, Colvin VL. 2013. Toxicity of Quantum Dots and Cadmium Salt to Caenorhabditis elegans after Multigenerational Exposure. *Environmental Science & Technology* 47:1148-1154.

[61] Gagne F, Auclair J, Turcotte P, Fournier M, Gagnon C, Sauve S, Blaise C. 2008. Ecotoxicity of CdTe quantum dots to freshwater mussels: Impacts on immune system, oxidative stress and genotoxicity. *Aquat Toxicol* 86:333-340.

[62] Kim J, Park Y, Yoon TH, Yoon CS, Choi K. 2010. Phototoxicity of CdSe/ZnSe quantum dots with surface coatings of 3-mercaptopropionic acid or tri-n-octylphosphine oxide/gum arabic in Daphnia magna under environmentally relevant UV-B light. *Aquat Toxicol* 97:116-124.

[63] Pace HE, Lesher EK, Ranville JF. 2010. INFLUENCE OF STABILITY ON THE ACUTE TOXICITY OF CdSe/ZnS NANOCRYSTALS TO DAPHNIA MAGNA. *Environ Toxicol Chem* 29:1338-1344.

[64] Wang J, Zhang X, Chen Y, Sommerfeld M, Hu Q. 2008. Toxicity assessment of manufactured nanomaterials using the unicellular green algae Chlamydonomas reinhardtii. *Chemosphere* 73:1121-1128.

[65] Zhang D, Hua T, Xiao F, Chen C, Gersberg RM, Liu Y, Ng WJ, Tan SK. 2014. Uptake and accumulation of CuO nanoparticles and CdS/ZnS quantum dot nanoparticles by Schoenoplectus tabernaemontani in hydroponic mesocosms. *Ecol Eng* 70:10.