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Dynamique, réactivité et écotoxicité des nanoparticules d'oxydes métalliques dans les sols : impact sur les fonctions et la diversité des communautés microbiennes

Marie Simonin

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Présentée par
Marie SIMONIN

Dynamique, réactivité et écotoxicité des nanoparticules d'oxydes métalliques dans
les sols : impact sur les fonctions et la diversité des communautés microbiennes

Thèse dirigée par Agnès RICHAUME-JOLION et Jean MARTINS

Soutenue le 12 octobre 2015 devant le jury composé de :

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Table des matières

Remerciements	1
Résumé	8
Abréviations	9
Introduction générale	10
Chapitre 1: Synthèse bibliographique, questions et choix méthodologiques	14
1. Les nanomatériaux, contaminants émergents des écosystèmes terrestres	15
a. Définitions.....	15
b. Utilisation des nanomatériaux dans des produits de consommation.....	16
c. Cycle de vie des nanomatériaux et rejet dans l'environnement.....	18
2. Devenir des nanomatériaux dans les sols	22
a. Transformations des nanomatériaux dans les sols	22
b. Transport des nanomatériaux dans les sols	26
3. Biodisponibilité et toxicité des nanomatériaux dans les sols	30
a. Biodisponibilité des nanomatériaux dans les sols	30
b. Toxicité des nanomatériaux dans les sols	32
4. Impact des nanomatériaux sur les communautés microbiennes du sol	34
a. « Impact of engineered nanoparticles on the activity, abundance, and diversity of soil microbial communities: a review”	34
5. Questions de la thèse	49
6. Méthodologie générale.....	50
a. Les nanoparticules d'oxydes métalliques.....	50
b. Les sols	50
c. Les dispositifs expérimentaux	51
d. Les indicateurs microbiens	52
Chapitre 2: Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques	55
1. Introduction	56
2. Influence des propriétés du sol sur le transport du TiO ₂ et CuO en colonnes de sol ..	57
a. Article 2 : Présentation générale de l'étude et synthèse des principaux résultats... 57	
b. «Influence of clay and organic matter content on the transport of TiO ₂ and CuO nanoparticles in saturated soil columns»	58

Table des matières

3.	Influence des propriétés du sol sur l'impact du TiO ₂ sur la respiration microbienne et l'abondance bactérienne	75
a.	Article 3 : Présentation générale de l'étude et synthèse des principaux résultats... 75	
b.	« Influence of soil properties on the toxicity of TiO ₂ nanoparticles on carbon mineralization and bacterial abundance »	76
4.	Conclusions du chapitre	86
Chapitre 3 : Impact du TiO₂ sur le cycle de l'azote : exemple d'un sol limoneux-argileux ..		88
1.	Introduction	89
2.	Effet d'une contamination aiguë sur le cycle de l'azote d'un sol	89
a.	Article 4 : Présentation générale de l'étude et synthèse des principaux résultats... 89	
b.	« Response of soil microbial community to titanium dioxide nanoparticles: a cascading pitch on the nitrogen cycle »	91
3.	Etude de l'effet dose-réponse du TiO ₂ sur l'activité nitrifiante et les groupes fonctionnels nitrifiants lors d'une contamination aiguë	123
a.	Article 5 : Présentation générale de l'étude et synthèse des principaux résultats. 123	
b.	« Soil nitrification is altered by titanium dioxide nanoparticles through modifications of coupling between ammonia- and nitrite-oxidizers »	125
4.	Comparaison des effets de contaminations aiguë et chronique sur le transport et la toxicité du TiO ₂ sur la nitrification	155
a.	Article 6 : Présentation générale de l'étude et synthèse des principaux résultats. 155	
b.	« Transport of TiO ₂ nanoparticles and toxicity on microbial communities under acute and chronic exposures in soil columns »	158
5.	Conclusions du chapitre	177
Chapitre 4 : Conclusions générales et perspectives		179
1.	Influence des propriétés du sol sur le devenir et l'écotoxicité des NPs d'oxydes métalliques	179
2.	Impact des TiO ₂ -NPs sur le cycle de l'N.....	183
3.	Effet d'une pollution chronique	187
4.	Une approche « systems biology » au service de l'écotoxicologie	189
Références		190

Table des illustrations

Figure 1: Schéma général d'organisation des résultats de la thèse.....	13
Figure 2: L'échelle des dimensions : du visible à l'invisible (INRS, 2015).....	16
Figure 3: Exemples d'utilisation des nanoparticules dans des domaines divers (Tsuzuki, 2009).....	17
Figure 4: Cycle de vie des nanomatériaux manufacturés dans l'environnement, ENM : Engineered Nanomaterial (Sun <i>et al.</i> , 2014).....	18
Figure 5: Transformations des nanomatériaux manufacturés au cours de leur utilisation et leur rejet dans l'environnement, ENM : Engineered Nanomaterial (Nowack <i>et al.</i> , 2012).....	19
Figure 6: Flux global des nanomatériaux en 2010 (en tonne par an) depuis leur production à leur utilisation et jusqu'à leur éventuelle élimination ou rejet dans l'environnement. Les différents stades du cycle de vie sont représentés de gauche (production) à droite (élimination ou rejet) (Keller <i>et al.</i> , 2014).....	21
Figure 7: Schéma de synthèse représentant les transformations chimiques (a), physiques (b), biologiques (c) et les interactions avec les macromolécules que les nanomatériaux subissent dans l'environnement, NOM : Natural Organic Matter, ROS : Reactive Oxygen Species, CNT : Carbon Nanotube (Lowry <i>et al.</i> , 2012).....	23
Figure 8: Schéma conceptuel représentant les transformations pouvant augmenter la similarité entre nanomatériaux ou leur diversité de forme, MNM : Manufactured Nanomaterial (Mitrano <i>et al.</i> , 2015).	26
Figure 9: Schématisation des processus clés déterminant le transport des colloïdes et nanomatériaux en milieu poreux: 1. Génération de colloïdes. 2. Lessivage des nanomatériaux. 3. Homoagrégation. 4. Dispersion. 5. Sédimentation. 6. Hétéroagrégation. 7. Exclusion de taille. 8. Filtration (<i>Straining</i>). 9. Déposition. 10. Transport convectif (Cornelis <i>et al.</i> , 2014).....	28
Figure 10: Influence de la concentration en oxalates d'aluminium et en carbone organique dissout sur la mobilité des Ag-NPs dans les sols (Cornelis <i>et al.</i> , 2013).....	29
Figure 11: Biodisponibilité et bioaccessibilité des contaminants dans les sols (Semple <i>et al.</i> , 2004).....	31
Figure 12: Schématisation des possibles mécanismes de toxicité des nanomatériaux sur des cellules bactériennes. CYP=cytochrome P (Handy <i>et al.</i> , 2008).....	32

Table des illustrations

Figure 13: Images de microscopie électronique à balayage (SEM-FEG, Zeiss, CMTC, G-INP) des nanoparticules de TiO_2 (à gauche) et de CuO (à droite) utilisées lors de la thèse.	50
Figure 14: Dispositif expérimental utilisé pour étudier le transport des nanoparticules dans des colonnes de sol (Photo : Marie Simonin). Les colonnes sont en verre et mesurent 1 cm de diamètre et 10 cm de long (Ge-Healthcare Pharmacia Biotech Inc.).	51
Figure 15: Incubation de microcosmes (flacon plasma 150 mL) contenant 50 g sol sec dans une étuve à 28°C (Photo : Marie Simonin).....	52
Figure 16: Schéma simplifié du cycle de l'azote dans les sols. Les groupes fonctionnels microbiens impliqués dans la nitrification et la dénitrification sont indiqués en rouge.	53
Figure 17: Découpage d'une colonne de sol en 4 profondeurs avant son incubation en microcosmes.	156

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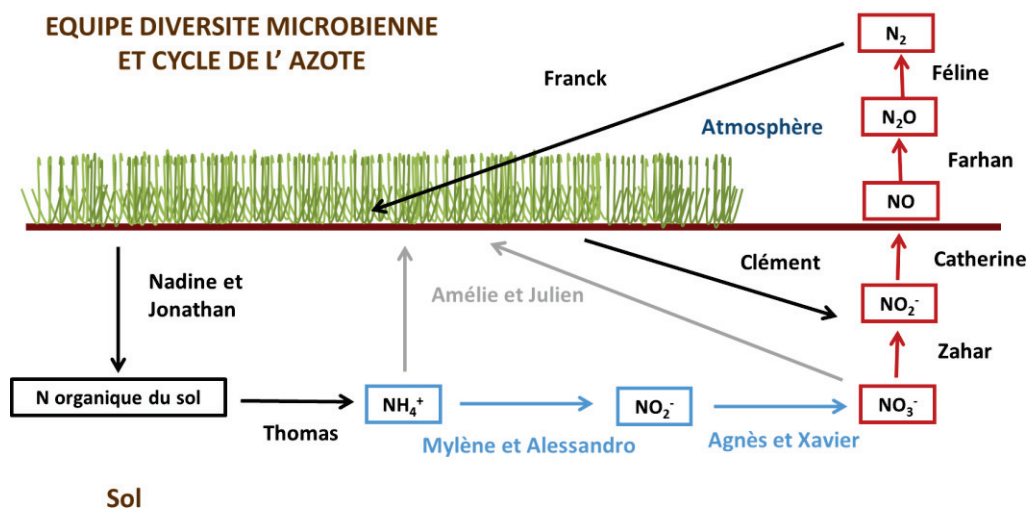
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Résumé

Les nanoparticules métalliques manufacturées (NPs) sont des polluants émergents dont la concentration augmente dans les sols en raison de leur utilisation croissante dans de nombreux produits commerciaux de la vie courante (cosmétiques, aliments, peintures...). Des études *in vitro* ont montré la toxicité des NPs pour les microorganismes, mais il existe encore peu de données sur l'écotoxicité et le devenir de ces contaminants dans les sols.

L'objectif de cette thèse est donc d'évaluer l'influence des paramètres abiotiques du sol sur (i) les caractéristiques physico-chimiques et le transfert des NPs, et (ii) sur la toxicité des NPs pour les communautés microbiennes du sol, en particulier pour des groupes fonctionnels microbiens impliqués dans le cycle du carbone et de l'azote.

Nous avons mis en évidence que les propriétés du sol influençaient l'agrégation et la charge de surface des NPs de dioxyde de titane (TiO_2) et d'oxyde de cuivre (CuO). Dans les six sols agricoles étudiés, nous avons observé un transport faible (CuO) à très faible (TiO_2) des NPs testées lors d'une expérimentation en colonnes de sols. Une étude en microcosmes de sol a permis de mettre en évidence une absence de toxicité des NPs de TiO_2 sur les communautés microbiennes, sauf dans un sol limono-argileux à forte teneur en matière organique. Dans ce sol, des effets négatifs ont été observés après 90 jours d'exposition sur les activités microbiennes (respiration, nitrification et dénitrification), sur l'abondance des microorganismes nitrifiants et la diversité des bactéries et des archées. De plus, nous avons montré que les effets négatifs observés sur la nitrification, même pour des concentrations extrêmement faibles de TiO_2 (0.05 mg kg^{-1}), étaient liés principalement à une forte sensibilité à ce polluant des archées oxydatrices de l'ammonium (AOA) impliquées dans ce processus. Des études complémentaires en colonnes de sol, ont permis de mettre en évidence des effets délétères des NPs plus importants sur la nitrification lors d'une contamination chronique au TiO_2 que lors d'une contamination aiguë.

Nous avons ainsi montré que la toxicité des NPs métalliques est réelle dans les sols et qu'elle est influencée par leurs propriétés physico-chimiques. Des processus microbiens clés comme la nitrification peuvent être fortement altérés par la présence de NPs. La rétention importante des NPs dans les sols suggère que ces polluants peuvent persister durablement dans les sols. Dans ce contexte, l'exposition chronique des sols sur le long terme doit être prise en compte dans des expérimentations visant à évaluer les risques environnementaux liés à la présence de ces polluants émergents.

Abréviations

AOA : Archées oxydatrices de l'ammonium

AOB : Bactéries oxydatrices de l'ammonium

Br : Brome (Bromide)

CCC : Concentration Critique de Coagulation

CEC : Capacité d'Echange Cationique

CNT : Carbon Nanotube

CuO : Oxyde de cuivre (copper oxide)

DCFH : 2',7'-dichlorofluorescein

DEA : Denitrification Enzyme Activity

DOC : Dissolved Organic Carbon

DLS : Dynamic Light Scattering

ICP-OES : Inductively Coupled Plasma - Optical Emission Spectrophotometer

MO : Matière Organique

MWCNT : Multi-walled carbon nanotube

NEA : Nitrification Enzyme Activity

NMDS : Non-Metric multi-Dimensional Scaling

NOB : Bactéries oxydatrices du nitrite

NP : Nanoparticule

OM : Organic Matter

OP : Oxidative potential

OTU : Operational Taxonomic Unit

PV : Pore Volume

qPCR : quantitative Polymerase Chain Reaction

ROS : Espèce réactive de l'oxygène (Reactive Oxygen Species)

SIR : Substrate-Induced Respiration

SWCNT : Single-Walled Carbon Nanotube

TiO₂ : Dioxyde de titane (titanium dioxide)

WHC: Water Holding Capacity

Introduction générale

Introduction générale

Les écosystèmes terrestres sont soumis en permanence à des perturbations d'origines anthropiques (e.g. pratiques agricoles, pollutions, compaction, salinisation). Les sols constituent un des principaux compartiments d'accumulation des polluants provenant d'origine diverses (irrigation, déversement accidentel, épandage de boues de station d'épuration, produits phytosanitaires, déposition atmosphérique...). De nouveaux types de contaminants qualifiés d'émergents, sont identifiés régulièrement dans les sols suite à la commercialisation de nouveaux produits pour lesquels il n'existe pas encore de réglementation et dont les effets environnementaux ne sont pas connus. Suite à l'avènement des nanotechnologies au début des années 2000, des inquiétudes ont émergé concernant le devenir et la toxicité des nanomatériaux dans l'environnement. En effet, les nanomatériaux présentent des propriétés exceptionnelles attirant l'intérêt des industriels (e.g. photo-catalytique, optique, semi-conducteur, antimicrobien). Cependant, ces propriétés peuvent, dans certains cas, leur conférer un potentiel toxique démontré en conditions *in vitro*. A ce jour, l'impact de ces polluants émergents a été étudié majoritairement dans les écosystèmes aquatiques, alors que les sols représentent vraisemblablement l'écosystème le plus exposé aux nanomatériaux.

En écotoxicologie, les sols reçoivent peu d'attention alors qu'ils rendent un très grand nombre de services écosystémiques clés parmi lesquels la production alimentaire, la qualité de l'eau et la régulation du climat qui font partie de grands enjeux socio-économiques actuels. Le sol est un système hétérogène complexe composé de fractions minérales, organiques, aqueuses et gazeuses. La complexité de l'étude des sols est également liée à la grande diversité de sols, en termes de texture, pH, ou encore de teneur et nature de matière organique qui sont autant de paramètres connus pour influencer le devenir et la toxicité des polluants. Dans une approche d'évaluation des risques, il est donc nécessaire de considérer l'influence des propriétés du sol sur les caractéristiques physico-chimiques du polluant, sur sa mobilité pour déterminer les risques de son transport et de sa dispersion en dehors du site de pollution initial, ainsi que sur sa biodisponibilité et sa toxicité.

Les sols hébergent de nombreux et divers organismes (macrofaune, mésofaune, microorganismes). Parmi eux les communautés microbiennes représentent une biodiversité considérable jouant un rôle fondamental dans les cycles biogéochimiques et la dégradation des polluants. Pour évaluer la qualité d'un sol et sa réponse à une perturbation, l'activité,

Introduction générale

l'abondance et la diversité des communautés microbiennes sont reconnues comme étant des indicateurs pertinents. En effet, les communautés microbiennes sont sensibles à de nombreuses perturbations qui peuvent altérer leur activité et par conséquent avoir des conséquences notables sur le fonctionnement de leur écosystème. De plus, cette approche permet d'étudier l'impact d'un contaminant à l'échelle d'une communauté d'organismes du sol et non pas uniquement sur une population bioindicatrice comme cela est fait plus classiquement en écotoxicologie.

De ce fait, afin d'évaluer les risques associés à des contaminants émergents tels que les nanomatériaux, une approche couplée abordant l'écodynamique du polluant dans le sol et son impact sur des indicateurs écologiques pertinents est nécessaire. C'est dans ce contexte qu'a été initiée cette thèse en lien avec un projet de recherche intitulé : « *Les nanoparticules métalliques dans les écosystèmes terrestres : écodynamique et impact sur les communautés bactériennes dans les sols* ». Ce projet a fait intervenir 2 laboratoires : Ecologie Microbienne (Lyon 1 - CNRS – INRA) et LTHE (Grenoble 1 – CNRS – G-INP - IRD) et a été financé par le programme national EC2CO (Ecosphère Continentale et Cotière) Microbien.

L'objectif principal de cette thèse était d'étudier l'influence des propriétés du sol sur la mobilité des nanoparticules de dioxyde de titane (TiO_2) et d'oxyde de cuivre (CuO) et leur toxicité sur des communautés microbiennes choisies en raison de leur rôle central dans la fertilité des sols. Les effets des nanoparticules sur des communautés microbiennes impliquées dans le cycle de l'azote (nitrifiants et dénitrifiants) ont donc été considérés afin d'évaluer la réponse de groupes fonctionnels microbiens présentant différents niveaux de redondance fonctionnelle. De plus, afin de déterminer l'influence du mode d'exposition sur le devenir et la toxicité des nanoparticules, des contaminations aiguës et chroniques ont été étudiées.

Ce travail est organisé en 4 chapitres :

- Le premier chapitre est une synthèse bibliographique concernant le devenir des nanomatériaux dans l'environnement et leur impact sur les communautés microbiennes qui fait l'objet d'un article publié. De cet état de l'art ont découlé les questionnements de la thèse et les choix méthodologiques pour mener à bien ce travail.

Introduction générale

- Dans le second chapitre sont présentés les résultats concernant l'influence des propriétés du sol sur le transport du TiO_2 et CuO , ainsi que sur la toxicité du TiO_2 sur les communautés microbiennes sous forme de deux articles.
- Dans le troisième chapitre sont présentés sous forme de trois articles, les résultats de l'impact du TiO_2 sur les communautés nitrifiantes et dénitrifiantes dans un sol limono-argileux lors de contaminations aiguës ou chroniques.
- Le quatrième chapitre est une conclusion générale sur l'ensemble des résultats obtenus. A la lumière de ces conclusions, les perspectives de recherche qui nous semblent les plus pertinentes sont proposées pour l'étude du devenir et de l'impact des nanomatériaux dans les sols.

La présentation des résultats de la thèse s'organise selon le schéma général présenté ci-dessous (Figure 1).

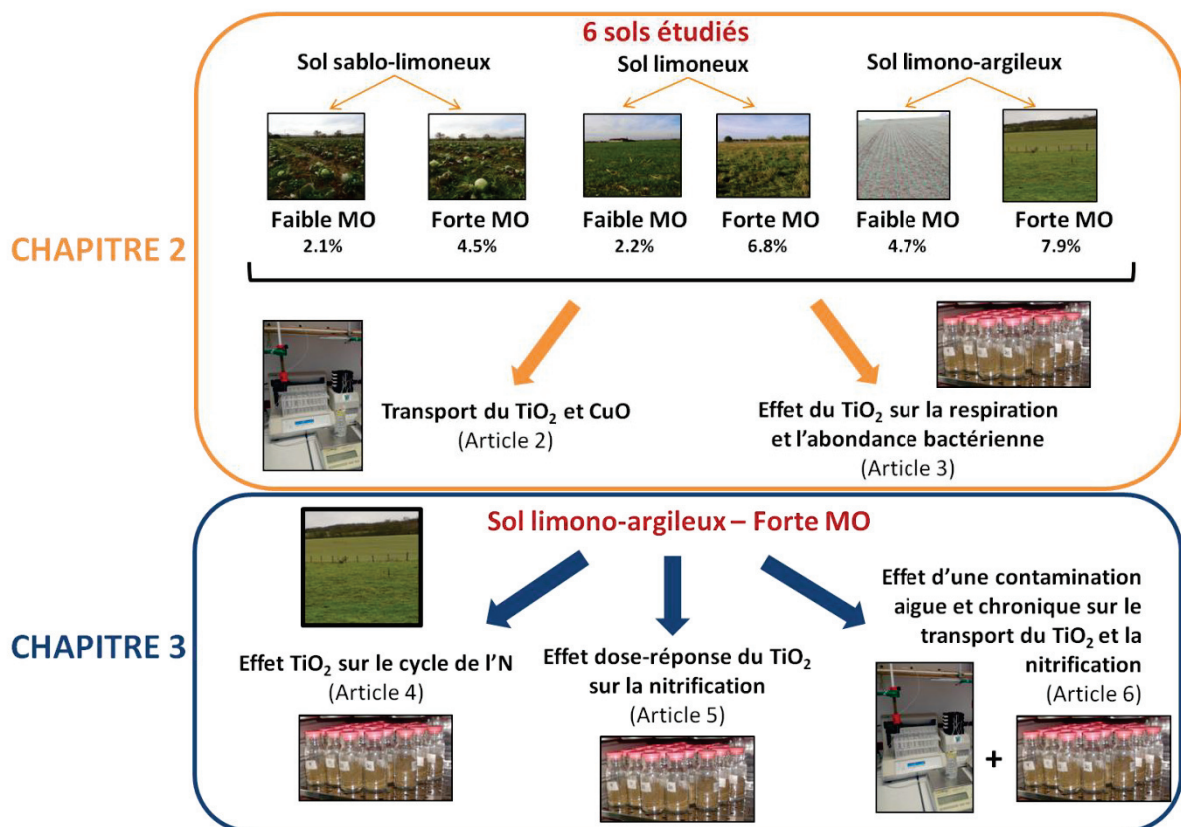


Figure 1: Schéma général d'organisation des résultats de la thèse.

Chapitre 1: Synthèse bibliographique, questions et choix méthodologiques

1. Les nanomatériaux, contaminants émergents des écosystèmes terrestres

a. Définitions

Un nanomatériau est défini comme *"un matériau naturel, formé accidentellement ou manufacturé contenant des particules libres, sous forme d'agrégat ou sous forme d'agglomérat, dont au moins 50 % des particules, dans la répartition numérique par taille, présentent une ou plusieurs dimensions externes se situant entre 1 nm et 100 nm"* (Figure 2, Commission européenne, 2011).

Parmi ces nanomatériaux, on distingue différentes familles de nano-objets manufacturés (INRS, 2012) :

- **les nanoparticules (NPs)** qui désignent des nano-objets dont les trois dimensions externes se situent à l'échelle nanométrique : nanoparticules de latex, d'oxyde de zinc, de fer et de cérium, d'alumine, de dioxyde de titane, de carbonate de calcium, etc ;
- **les nanofibres, nanotubes, nanofilaments ou nanobâtonnets** qui se rapportent à des nano-objets dont deux dimensions externes sont à l'échelle nanométrique et la troisième dimension significativement supérieure. Ces termes désignent des nano-objets longilignes de section comprise entre 1 et quelques dizaines de nm et de longueur comprise entre 500 et 10 000 nm comme, par exemple, les nanotubes de carbone, les nanofibres de polyester, les nanotubes de bore, etc...;
- **les nano-feuillets, nano-plats ou nano-plaquettes** qui définissent des nano-objets dont une dimension externe se situe à l'échelle nanométrique, les deux autres étant significativement supérieures. C'est le cas des nano-feuillets d'argile, des nano-plaquettes de séléniure de cadmium.

Les nanomatériaux peuvent donc se présenter sous un grand nombre de formes différentes, mais ils sont également caractérisés par une grande variété de propriétés chimiques. On peut distinguer les nanomatériaux en fonction de la nature organique ou inorganique de l'élément chimique les constituant. Les nanomatériaux organiques sont majoritairement commercialisés sous forme de nanotubes de carbone ou de fullerènes. Les nanomatériaux

inorganiques sont divisés en 3 classes : les nanomatériaux métalliques (argent, or...), les oxydes métalliques (titane, fer, cérium, cuivre...), et les quantum-dots (séléniure de cadmium) (Ju-Nam and Lead, 2008).

Les nanomatériaux ne peuvent donc pas être considérés comme un seul groupe homogène de contaminants, et leur diversité doit être prise en compte pour l'évaluation des risques liés à ces composés qui sont incorporés dans de nombreux produits industriels et commerciaux depuis une vingtaine d'années.

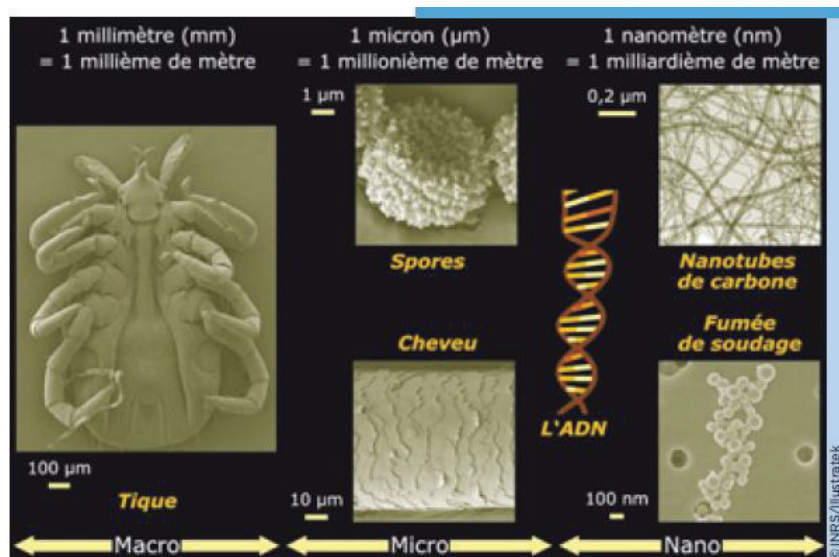


Figure 2: L'échelle des dimensions : du visible à l'invisible (INRS, 2015)

b. Utilisation des nanomatériaux dans des produits de consommation

Les nanomatériaux sont caractérisés par un pourcentage élevé d'atomes en surface ce qui leur confèrent des propriétés nouvelles et une forte réactivité comparées à celles des matériaux de même composition chimique mais de plus grande taille (Auffan *et al.*, 2009). Des propriétés électriques, optiques ou catalytiques par exemple, sont ainsi modifiées à l'échelle nanométrique. Ceci accorde aux nanomatériaux des propriétés physico-chimiques d'intérêt dans le milieu industriel et explique que l'ère des nanotechnologies est souvent décrite comme ouvrant une nouvelle révolution industrielle. Le marché des nanomatériaux est en effet estimé à 1 milliard de dollars en 2015 (Nel *et al.*, 2006).

Chapitre 1: Synthèse bibliographique, questions et choix méthodologiques

Les nanomatériaux manufacturés sont déjà utilisés dans des domaines très variés tels que l'agriculture, l'automobile, les cosmétiques, la construction, l'énergie, l'électronique, le médical, les peintures, les produits alimentaires, la remédiation, ou encore les textiles (Keller *et al.*, 2013; Mitrano *et al.*, 2015) (Figure 3). Actuellement on estime que la NP la plus utilisée dans l'industrie est le dioxyde de titane (TiO₂) (Robichaud *et al.*, 2009; Sun *et al.*, 2014). Ses propriétés photo-catalytiques exceptionnelles permettent en effet son utilisation dans de très nombreux secteurs d'activités (Chen and Mao, 2007). Cependant, le constat de cette utilisation croissante des nanomatériaux dans des produits de la vie courante implique la connaissance de leur cycle de vie mais aussi celle de leur devenir dans l'environnement.

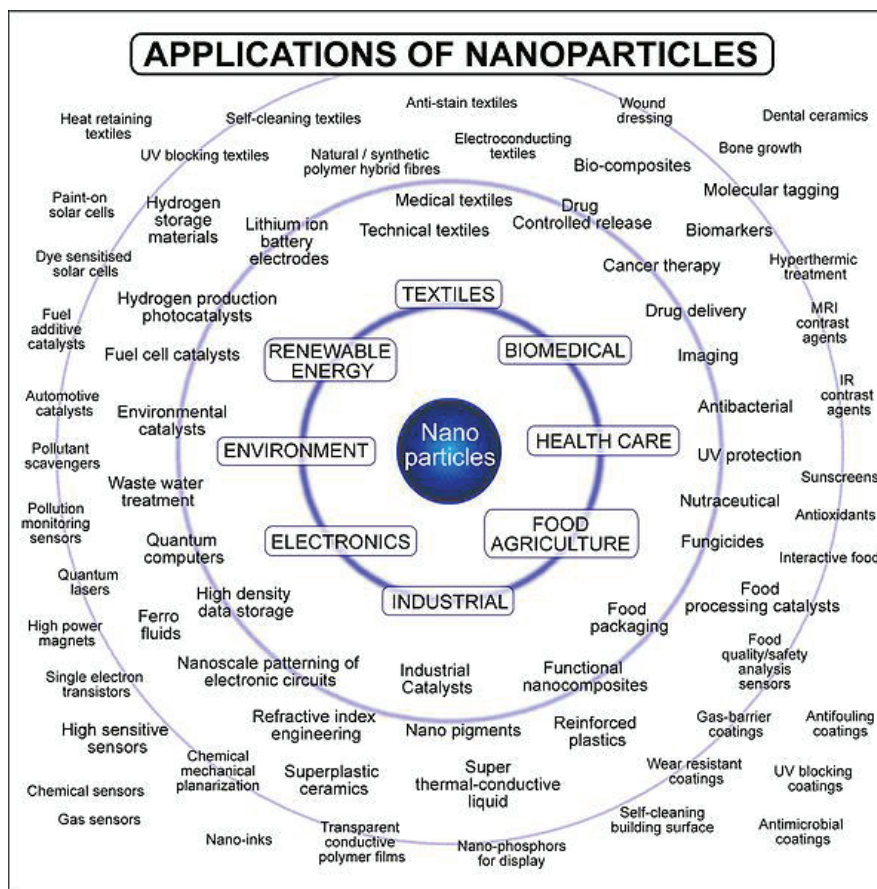


Figure 3: Exemples d'utilisation des nanoparticules dans des domaines divers (Tsuzuki, 2009).

c. Cycle de vie des nanomatériaux et rejet dans l'environnement

L'utilisation croissante de nanomatériaux soulève des interrogations quant à leur rejet dans l'environnement au cours des différentes étapes de leur cycle de vie, depuis la production jusqu'au retraitement (Figure 4).

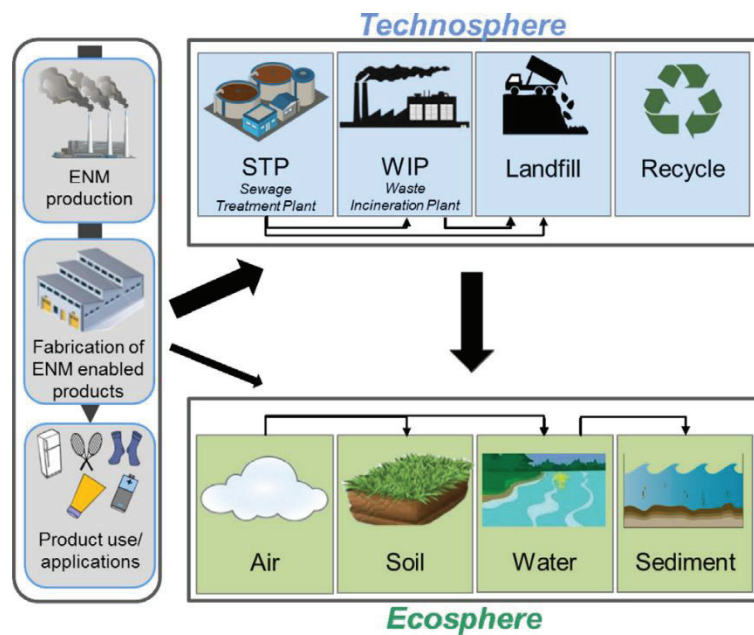


Figure 4: Cycle de vie des nanomatériaux manufacturés dans l'environnement, ENM : Engineered Nanomaterial (Sun *et al.*, 2014)

Som *et al.*, (2010) ont souligné l'importance de considérer le cycle de vie des nanomatériaux pour chaque produit afin d'obtenir une évaluation fiable des risques environnementaux associés. En effet, la forme sous laquelle seront rejetés les nanomatériaux dépendra de l'étape du cycle de vie du produit concerné (Nowack *et al.*, 2012) : (i) purs lors de la production, (ii) associés à d'autres composés lors de leur utilisation, (iii) associés à des composés environnementaux lors de leur entrée dans les écosystèmes (Figure 5).

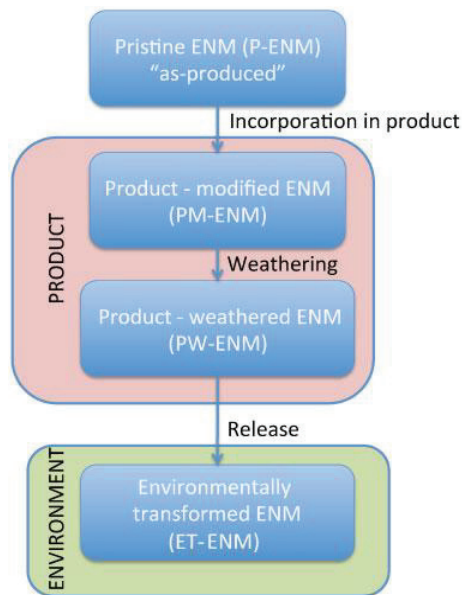


Figure 5: Transformations des nanomatériaux manufacturés au cours de leur utilisation et leur rejet dans l'environnement, ENM : Engineered Nanomaterial (Nowack *et al.*, 2012).

Dans ce contexte, de nombreuses études ont été conduites pour étudier le vieillissement et les transformations des nano-produits au cours de leur cycle de vie (Mitrano *et al.*, 2015). Plusieurs travaux ont mis en évidence que des nanomatériaux pouvaient être rejetés de manière involontaire dans l'environnement suite à leur utilisation dans des produits couramment utilisés comme les peintures, les textiles ou les cosmétiques (Labille *et al.*, 2010; Nowack *et al.*, 2012). Le relargage de TiO₂-NPs à partir des peintures appliquées sur des façades extérieures a été montré par Kaegi *et al.*, (2008) qui ont mis en évidence un transfert de concentrations importantes ($3,5 \cdot 10^7$ TiO₂ particules/L de taille inférieure à 100 nm) vers les eaux de surfaces dans des conditions météorologiques naturelles. Il a été également estimé que 75 à 95% des nanomatériaux contenus dans les cosmétiques et crèmes solaires (e.g. TiO₂ et ZnO) étaient libérés dans l'environnement suite à une douche ou une baignade (Keller *et al.*, 2013). Ces différents exemples suggèrent que les nanomatériaux entrent involontairement dans les écosystèmes aquatiques et terrestres où ils constituent des pollutions chroniques.

Il existe également des cas où l'environnement est exposé de manière volontaire à des nano-produits, en particulier dans le cadre d'activités liées à l'agriculture ou à la remédiation des écosystèmes pollués. En effet, en agronomie de plus en plus de nanofertilisants et nanopesticides sont proposés sur le marché pour augmenter les rendements de production

Chapitre 1: Synthèse bibliographique, questions et choix méthodologiques

végétale et lutter contre des phytopathogènes (Liu and Lal, 2015; Servin *et al.*, 2015). Les nanomatériaux présentent également une grande efficacité pour la remédiation des écosystèmes contaminés aux métaux lourds, polluants organiques et substances biologiques (virus, bactéries, antibiotiques). Ils sont ainsi utilisés pour le traitement des eaux usées, des déchets industriels ou de l'air (Hua *et al.*, 2012; Khin *et al.*, 2012).

A cause de limitations techniques, il est encore très difficile de détecter les nanomatériaux et d'en mesurer les concentrations dans des milieux environnementaux complexes (Tourinho *et al.*, 2012; Cornelis *et al.*, 2014). Cependant, comme décrit ci-dessus, il existe des preuves que les nanomatériaux sont déjà présents dans de nombreux environnements et que leurs concentrations environnementales augmentent au cours du temps.

Tableau 1: Prédiction des concentrations en TiO₂-NPs dans différents compartiments environnementaux dans les pays de l'union européenne (Sun *et al.*, 2014).

Environnement	Concentration en TiO ₂ prédite
Air	0.001 µg m ⁻³
Eaux de surface	0.53 µg L ⁻¹
Sédiments	1.9 mg kg ⁻¹ par an
Boues de station d'épuration	170 mg kg ⁻¹
Sol naturel et urbain	0.13 µg kg ⁻¹ par an
Sol traité avec boues de station d'épuration	1.2 mg kg ⁻¹ par an

Sur la base de données de production annuelle de nanomatériaux et du cycle de vie des produits, des études ont permis de modéliser les concentrations environnementales des nanomatériaux les plus couramment utilisés. Plusieurs études concordent sur le fait que le nanomatériau ayant la plus forte concentration dans les différents compartiments de la technosphère et de l'écosphère est le TiO₂ (Tableau 1 et Figure 6). De plus, ces travaux montrent que les sols et les sites de stockage constituent les réceptacles environnementaux majoritaires alors qu'une fraction beaucoup plus faible est rejetée dans les écosystèmes aquatiques et dans l'air (Gottschalk *et al.*, 2009; Keller *et al.*, 2013; Sun *et al.*, 2014).

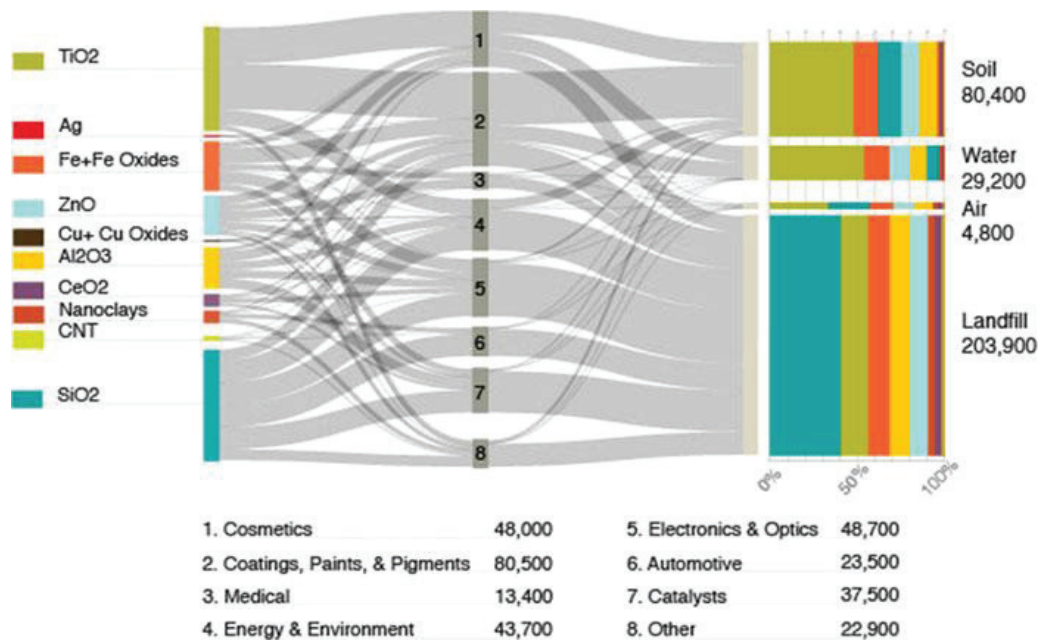


Figure 6: Flux global des nanomatériaux en 2010 (en tonne par an) depuis leur production à leur utilisation et jusqu'à leur éventuelle élimination ou rejet dans l'environnement. Les différents stades du cycle de vie sont représentés de gauche (production) à droite (élimination ou rejet) (Keller *et al.*, 2014).

Les nanomatériaux peuvent atteindre les sols via les eaux usées utilisées pour l'irrigation et surtout à travers l'épandage de boues de stations d'épuration utilisées comme fertilisants des champs agricoles dans certains pays, dont la France (Kiser *et al.*, 2009; Sun *et al.*, 2014; Yang *et al.*, 2014). En effet, il est estimé que 90% des NPs d'argent (Ag), de ZnO, TiO₂ et de cérium (Ce) sont retenues dans les boues lors du retraitement des eaux usées en station d'épuration (Cornelis *et al.*, 2014). Les applications directes de nano-produits sous forme de nanofertilisants ou nanopesticides sont encore très peu prises en compte dans les estimations, du fait de leur apparition récente sur le marché. De plus, l'estimation de la production mondiale de nanomatériaux est encore mal évaluée (Piccinno *et al.*, 2012). Il est donc très probable que les concentrations prédites par ces modèles sous-estiment encore fortement les concentrations réelles dans certains sols agricoles.

2. Devenir des nanomatériaux dans les sols

a. Transformations des nanomatériaux dans les sols

Le fort ratio aire de surface / volume des nanomatériaux entraîne une forte réactivité et des propriétés physico-chimiques très dynamiques dans un contexte environnemental (Lowry *et al.*, 2012). De ce fait, ces polluants émergents peuvent subir de nombreuses transformations lorsqu'ils sont en contact avec les composés et les organismes du sol. Ces transformations sont susceptibles d'affecter leur devenir, leur mobilité et leur toxicité dans les sols (Pan and Xing, 2012). Ces aspects doivent donc être étudiés afin d'améliorer l'évaluation des risques associés à ces contaminants.

Les nanomatériaux peuvent subir 4 grands types de transformations dans l'environnement (Figure 7) : (i) chimiques, (ii) physiques, (iii) biologiques, et (iv) des interactions avec des macromolécules. Ces différents processus peuvent conduire à des modifications d'agrégation (homo- et hétéroagrégation), à la dissolution de certains nanomatériaux et au vieillissement des enrobages (coating) qui peuvent leur être associés.

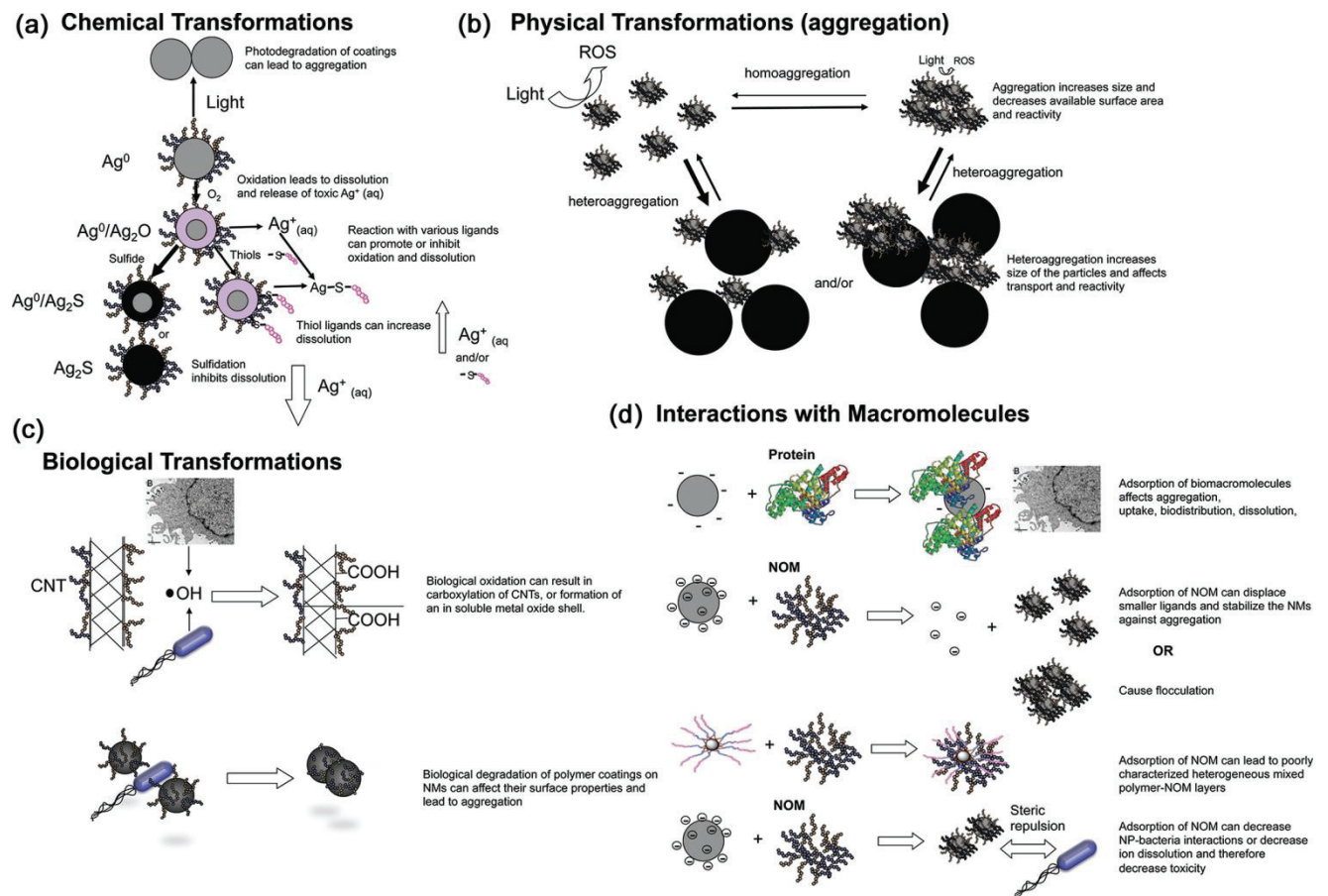


Figure 7: Schéma de synthèse représentant les transformations chimiques (a), physiques (b), biologiques (c) et les interactions avec les macromolécules que les nanomatériaux subissent dans l'environnement, NOM : Natural Organic Matter, ROS : Reactive Oxygen Species, CNT : Carbon Nanotube (Lowry *et al.*, 2012).

En raison de limitations techniques, les transformations des nanomatériaux ne peuvent pas être étudiées directement dans les matrices environnementales complexes comme les sols. Actuellement, les techniques disponibles pour caractériser les nanomatériaux sur la base de leur taille, de leur charge et de leur composition chimique ne sont applicables qu'en phase aqueuse (Dynamic Light Scattering, microscopie électronique à transmission et balayage, microscope à force atomique, EXAFS...) (Tourinho *et al.*, 2012). C'est pourquoi les données disponibles dans la littérature ont été majoritairement acquises à partir de milieux aqueux simplifiés dans lesquels des paramètres physico-chimiques environnementaux d'intérêt ont été modifiés.

Plusieurs facteurs environnementaux sont bien connus pour modifier l'homoagrégation (agrégation entre nanomatériaux, Figure 7b), les plus étudiés étant le pH et la force ionique (Cornelis *et al.*, 2014). En particulier, ces facteurs influencent les charges de surface qui

peuvent être déterminées via la mesure du potentiel zêta (mobilité électrophorétique). Une forte homoagrégation est observée au pH proche du point isoélectrique du nanomatériau (i.e. au pH pour lequel la charge de surface nette est égale à zéro) et également pour une force ionique élevée jusqu'à la concentration critique de coagulation (CCC). Dans ce cas, l'agrégation n'augmente plus à cause de la diffusion (Darlington *et al.*, 2009; Thio *et al.*, 2011; Cornelis *et al.*, 2014). Au vu des valeurs de force ionique retrouvées dans les sols, très en deçà des CCC mesurées, il est prédit que les nanomatériaux aient un faible taux d'homoagrégation dans les sols, excepté aux pH proches du point isoélectrique des NPs considérées (Cornelis *et al.*, 2014).

Le processus physique susceptible d'être le plus courant dans les pores du sol est l'hétéroagrégation (i.e. agrégation avec des colloïdes naturels autres que les nanomatériaux) (Figure 7b et 7d, Cornelis *et al.*, 2011; Zhou *et al.*, 2012). Cette hypothèse est supportée par le fait que les hétéroagrégats de colloïdes naturels sont beaucoup plus fréquents dans les sols que les homoagrégats (Buffle *et al.*, 1998) et que la concentration en particules environnementales est beaucoup plus grande que celle des nanomatériaux. L'hétéroagrégation est un phénomène beaucoup plus difficile à étudier expérimentalement que l'homoagrégation, particulièrement dans les sols car il existe une très grande variété de colloïdes ou particules (matière organique, argile, limon...) pouvant potentiellement interagir avec les nanomatériaux. Des études pionnières se sont intéressées aux processus d'hétéroagrégation et ont montré que la matière organique (MO) dissoute pouvait s'adsorber à la surface des TiO₂-NPs leur conférant un potentiel zêta négatif (Zhang *et al.*, 2009). En milieu aquatique, les argiles de type montmorillonite entraînent une déstabilisation des NPs en suspension, qu'elles soient chargées positivement ou négativement (Ag et TiO₂) et les interactions NPs-argile sont dépendantes du pH et de la force ionique (Zhou *et al.*, 2012). Cornelis *et al.*, (2011) ont montré que du phosphate dissous pouvait s'adsorber en surface de CeO₂-NPs, mais aussi qu'une hétéroagrégation pouvait avoir lieu avec des argiles dans des conditions représentatives du sol. Dans une étude associée, les Ag-NPs chargées négativement s'adsorbaient préférentiellement sur les sites de surface en bordure des feuillettes d'argile (oxyde de fer ou d'aluminium) qui sont chargés positivement (Cornelis *et al.*, 2011). Les transformations subies par les nanomatériaux dans les sols demeurent néanmoins très difficilement prévisibles. De plus,

elles ne peuvent être généralisées sur la base de ce qui est connu pour un sol donné en raison des caractéristiques propres aux différents types d'argiles, de la typologie des MO, des conditions de pH et de la force ionique qui varient d'un sol à un autre.

En plus des processus d'homo- et hétéroagrégation, certains nanomatériaux peuvent aussi subir une dissolution (Figure 7a). Les nanomatériaux constitués de métaux de classe B (e.g. Ag, Zn) sont plus susceptibles de se solubiliser dans l'environnement car ils forment des oxydes métalliques partiellement solubles et ont une forte affinité avec les ligands inorganiques et organiques sulfurés (Lowry *et al.*, 2012). La dissolution des nanomatériaux entraîne le relargage d'ions métalliques connus pour être toxiques vis à vis des organismes du sol. La compréhension de cette toxicité requiert la connaissance du taux de dissolution dans les sols et de la concentration relative des formes dissoutes et nanoparticulaires au cours du temps (Tourinho *et al.*, 2012). Toutefois, ces données sont encore très limitées à cause des difficultés techniques pour séparer les fractions dissoutes et nanoparticulaires à partir des sols. Vittori Antisari *et al.*, (2013) ont observé une faible dissolution des CeO₂, Fe₃O₄ et SnO₂-NPs dans un sol de texture équilibrée. Dans 16 sols différents, aucune dissolution des CeO₂-NPs n'a été constatée (Cornelis *et al.*, 2011). En revanche, des Ag et ZnO-NPs peuvent se solubiliser de manière significative (Kool *et al.*, 2011; Shoults-Wilson *et al.*, 2011; Rousk *et al.*, 2012; Diez-Ortiz *et al.*, 2015), mais ce processus est limité par la présence d'un enrobage (Coutris *et al.*, 2012). La dissolution des NPs semblant être un processus qui s'effectue sur le long terme dans les sols, il serait nécessaire d'étudier le vieillissement des NPs sur des longues durées et d'évaluer leur toxicité dans ces conditions (Coutris *et al.*, 2012; Whitley *et al.*, 2013; Diez-Ortiz *et al.*, 2015).

La modification des nanomatériaux suite aux activités biologiques est inévitable dans les sols en raison de la production de métabolites par les plantes et les microorganismes ou à l'ingestion par la faune. Ces transformations, qui concernent les nanomatériaux purs ou enrobés, sont notamment associées aux réactions redox en lien avec l'activité biologique (Lowry *et al.*, 2012). Il est vraisemblable que la biotransformation la plus courante soit l'adsorption de biomacromolécules (e.g. protéines, polysaccharides) à la surface des nanomatériaux (forme d'hétéroagrégation, Figure 7c). La biodégradation des nanomatériaux à base de carbone est possible mais dépend du type de sol et des enzymes disponibles dans les pores du sol (Petersen *et al.*, 2011). Il a également été montré que les enrobages des

nanomatériaux utilisés pour limiter leur homoagrégation et favoriser leur stabilité, comme le citrate, peuvent être facilement dégradés, en particulier par des espèces réactives de l'oxygène (Reactive Oxygen Species – ROS) produites par des champignons (Navarro *et al.*, 2011).

Les nanomatériaux peuvent donc subir des transformations multiples et complexes dans les sols qui vont modifier leur agrégation, leur charge de surface et leur réactivité et donc possiblement affecter leur devenir et toxicité. Face à cette complexité, des tentatives sont faites pour classer les différents types de transformations subies par les NPs afin de limiter le nombre d'expérimentations à réaliser et permettre une meilleure évaluation des risques (Mitrano *et al.*, 2015, Figure 8).

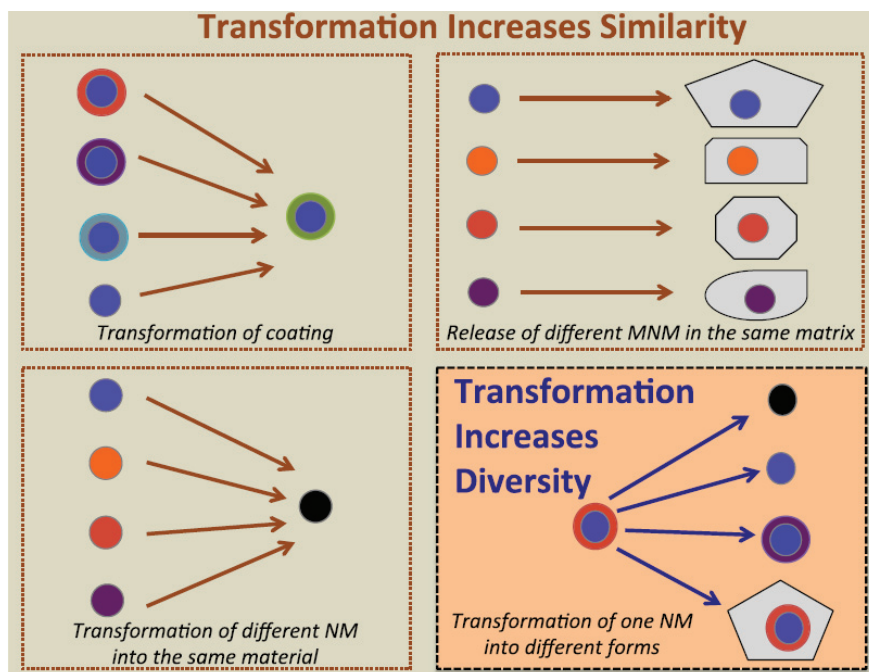


Figure 8: Schéma conceptuel représentant les transformations pouvant augmenter la similarité entre nanomatériaux ou leur diversité de forme, MNM : Manufactured Nanomaterial (Mitrano *et al.*, 2015).

b. Transport des nanomatériaux dans les sols

Depuis une dizaine d'années, un grand nombre d'études a été consacré au transport des nanomatériaux en milieu poreux afin de prédire leur devenir dans les eaux souterraines et les sols. La majorité de ces travaux a été conduite dans des situations simplifiées en utilisant des colonnes représentant des structures porales homogènes (sable ou billes de verre) en

conditions de saturation en eau. Cette littérature se base sur les bonnes connaissances du transport des colloïdes en milieu poreux (e.g. Vitorge, 2010; Elimelech *et al.*, 2013) associées en particulier à la théorie Derjaguin-Landau-Verwey-Overbeek (DLVO). Les différents processus déterminant le transport des colloïdes et des nanomatériaux en milieu poreux sont représentés dans la figure 8.

En utilisant des systèmes simplifiés, plusieurs facteurs environnementaux affectant la mobilité des nanomatériaux ont été identifiés, en particulier la force ionique de la solution, le pH, la présence d'acides organiques, la concentration et la taille des nanomatériaux ou la vitesse de l'eau (Lecoanet *et al.*, 2004; Espinasse *et al.*, 2007; Darlington *et al.*, 2009; Solovitch *et al.*, 2010; Wang *et al.*, 2012). Le transport des nanomatériaux augmente fortement lorsque ceux-ci sont stables en suspension (i.e. ne sédimentent pas) (Wang *et al.*, 2012). Ces conditions sont réunies lorsqu'ils sont faiblement agrégés (taille < 300 nm), que la force ionique de la solution est faible, en présence d'acides humiques ou fulviques, ou de surfactants. La vitesse de l'eau est également à prendre en compte : généralement, plus le flux d'eau est important, plus le transport de NPs est important (Lecoanet and Wiesner, 2004; Chowdhury *et al.*, 2012; Jeong and Kim, 2009). De manière générale, il a été observé que la taille des nanomatériaux augmentait suite à leur transport dans la colonne suggérant des processus d'homo- et d'hétéroagrégation (Figure 9 – processus 3 et 6, (Fang *et al.*, 2009; Wang *et al.*, 2012). Logiquement, les nanomatériaux de plus petite taille ont des capacités de transport plus grandes (Darlington *et al.*, 2009; Ben-Moshe *et al.*, 2010). Darlington *et al.*, (2009) ont montré que les NPs de 200nm avaient une mobilité beaucoup plus importante que les agrégats de plus grande taille due au processus de filtration et à la sédimentation (Figure 9 – processus 5 et 8). Différentes études ont mis en évidence que la mobilité des NPs variait fortement d'un nanomatériau à un autre (Lecoanet and Wiesner, 2004; Lecoanet *et al.*, 2004; Ben-Moshe *et al.*, 2010; Tian *et al.*, 2010). Les nanomatériaux à base de carbone (fullerène, SWCNT, MWCNT) semblent particulièrement peu mobiles comparés aux nanomatériaux métalliques, sans doute à cause de leur très forte hydrophobicité qui empêche leur dispersion dans l'eau et donc leur mobilité (Jaisi *et al.*, 2008; Tian *et al.*, 2012).

Ces données obtenues en conditions simplifiées apportent des informations importantes pour la compréhension du devenir des nanomatériaux dans les sols. Cependant, elles ne représentent pas de manière satisfaisante la variété des types de surfaces minérales et

organiques, l'hétérogénéité des charges de surface, la granulométrie et la chimie de la solution des sols (Fang *et al.*, 2009).

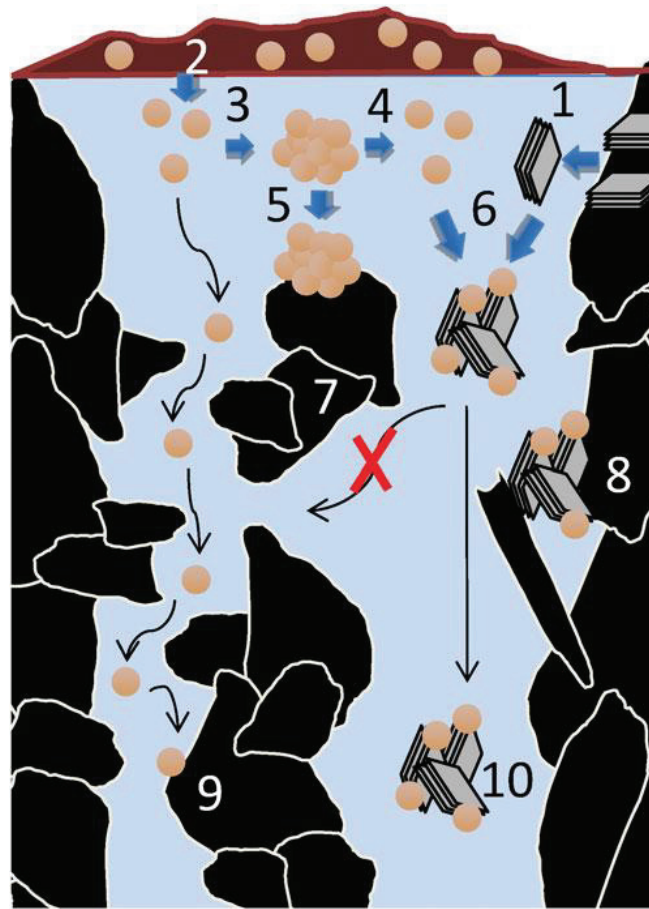


Figure 9: Schématisation des processus clés déterminant le transport des colloïdes et nanomatériaux en milieu poreux: 1. Génération de colloïdes. 2. Lessivage des nanomatériaux. 3. Homoagrégation. 4. Dispersion. 5. Sédimentation. 6. Hétéroagrégation. 7. Exclusion de taille. 8. Filtration (*Straining*). 9. Déposition. 10. Transport convectif (Cornelis *et al.*, 2014).

Un nombre limité de publications concernant le transfert des nanomatériaux dans les sols naturels est disponible à l'heure actuelle (Sun *et al.*, 2015). Il a été montré que les ZnO-NPs ont une mobilité plus faible dans un sol que dans du sable. Par ailleurs, les effets de la concentration de NPs et de la force ionique étaient très différents dans ces 2 types de milieux. Dans le sable, le transport du ZnO diminuait lorsque la concentration appliquée augmentait, alors que le phénomène inverse était observé dans le sol (Sun *et al.*, 2015). Deux études portant sur le transport des TiO₂-NPs ont montré, soit aucun transport dans les 3 sols testés (Nickel *et al.*, 2015), soit une mobilité dans 8 sols sur les 12 étudiés (Fang *et al.*, 2009). Dans cette dernière étude, le transport du TiO₂ était influencé par la teneur en argile et la force ionique de la solution du sol. Dans ces 2 études, les concentrations, la préparation

des colonnes et les protocoles de minéralisation étaient très différents, ce qui a pu influencer de manière importante sur les résultats obtenus. Nickel *et al.*, (2015) ont tenté d'appliquer les protocoles standardisés de l'OCDE TG 312 « Lessivage en colonne de sol » et suggèrent que ce protocole doit être adapté lorsque l'on utilise des nanomatériaux, en particulier le TiO_2 .

Un transport limité des nanomatériaux à base de carbone dans les sols (fullerènes, SWCNT et MWCNT) a également été rapporté (Jaisi and Elimelech, 2009; Kasel *et al.*, 2013). Le même constat a été fait pour les quantum dots de CdSe et CdSe/ZnS, dont la majorité était retenue à l'entrée de la colonne (Navarro *et al.*, 2011). Ces auteurs ont montré toutefois que la mobilité des quantum dots était significativement augmentée en présence d'acides organiques au pouvoir chélatant (comme l'EDTA), ces derniers pouvant être présents en quantité importante dans les sols, notamment via l'exsudation racinaire. Cornelis *et al.*, (2013) ont observé le transport d'Ag-NPs dans 8 sols sur les 11 testés, en appliquant une concentration très faible de 1.7 mg Ag L^{-1} . Le transfert des Ag-NPs était cependant assez faible (91 à 98.8% retenus dans le sol), alors que les sols utilisés étaient très sableux (60 à 99% de sable). La faible mobilité des Ag-NPs a été expliquée par une forte hétéroagrégation entraînant des processus d'exclusion de taille, de filtration et de déposition sur des oxalates d'aluminium positivement chargés (Cornelis *et al.*, 2013, Figure 10, Figure 9 –processus 7, 8 et 9).

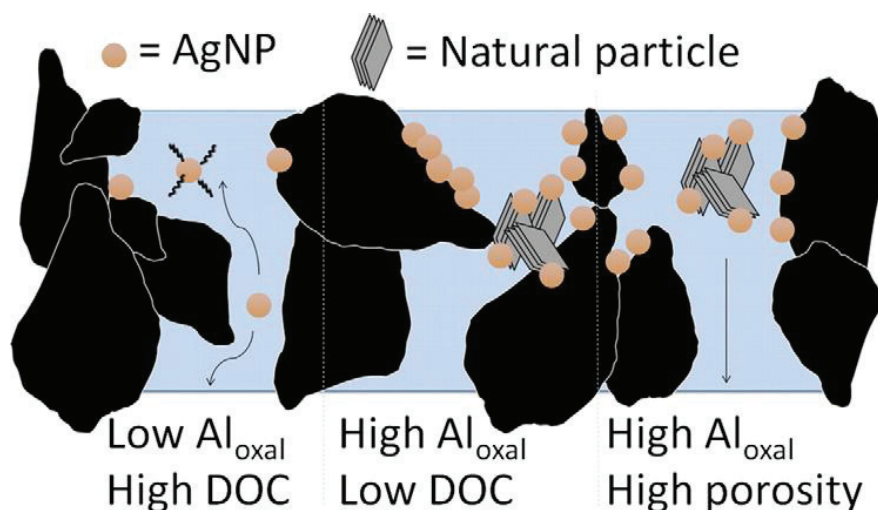


Figure 10: Influence de la concentration en oxalates d'aluminium et en carbone organique dissout sur la mobilité des Ag-NPs dans les sols (Cornelis *et al.*, 2013)

Toutes, ces études pionnières suggèrent que les nanomatériaux ont une très faible mobilité dans les sols naturels et donc qu'il existe un faible risque de transport de ces contaminants

hors du site pollué, que ce soit verticalement vers les eaux souterraines ou latéralement vers des sites voisins. Cependant, ces conclusions restent à valider en raison du faible nombre d'études disponibles (6 types de NPs étudiés pour 12 publications au total) et des nombreuses questions qui subsistent. En effet, la plupart des études disponibles ont été conduites en conditions de saturation en eau, avec de fortes concentrations en nanomatériaux, des méthodologies variées en termes de préparation des suspensions de nanomatériaux et en utilisant principalement des sols de type sableux. A l'heure actuelle, les paramètres clés du sol influençant le transport des nanomatériaux ne sont pas identifiés, alors que ces informations permettraient une meilleure évaluation des risques suite à des épandages de boues ou l'utilisation de nanofertilisants sur un sol. De plus, il apparaît nécessaire d'étudier la mobilité des nanomatériaux dans des conditions plus réalistes d'exposition en considérant par exemple des apports via l'épandage de boues, des contaminations chroniques, l'application de faibles concentrations ou l'influence de la végétation.

3. Biodisponibilité et toxicité des nanomatériaux dans les sols

a. Biodisponibilité des nanomatériaux dans les sols

Les nanomatériaux peuvent avoir des temps de résidence longs dans les sols et subir de nombreuses transformations. Les plantes et les organismes du sol (microorganismes, mésofaune et macrofaune) seront donc exposés à ces contaminants émergents. Dans ce contexte, le point clé est de déterminer comment les organismes sont exposés aux nanomatériaux présents dans les différents compartiments du sol, qu'il s'agisse de la solution du sol ou des agrégats et quelle est la fraction biodisponible de ces contaminants en fonction des organismes étudiés (Tourinho *et al.*, 2012). Selon la définition de Semple *et al.*, (2004), la biodisponibilité d'un composé dans les sols est définie de la manière suivante en écotoxicologie : « *un composé biodisponible est librement disponible pour traverser la membrane cellulaire d'un organisme à partir du milieu où l'organisme se trouve à un temps donné* » et la bioaccessibilité est définie ainsi : « *le composé bioaccessible est librement disponible pour traverser la membrane cellulaire d'un organisme depuis l'environnement, si l'organisme a accès à ce composé. Toutefois, le composé peut être soit éliminé physiquement de l'organisme ou seulement biodisponible après un certain temps* ». D'après ces définitions,

la bioaccessibilité englobe donc ce qui est actuellement biodisponible et ce qui est susceptible de le devenir (Figure 11, Semple *et al.*, 2004). Dans les sols, on s'attend donc à ce que la concentration biodisponible en nanomatériaux soit plus faible que la concentration totale, du fait du grand nombre de surfaces réactives pouvant immobiliser les nanomatériaux et de transformations pouvant affecter leur biodisponibilité (Cornelis *et al.*, 2014).

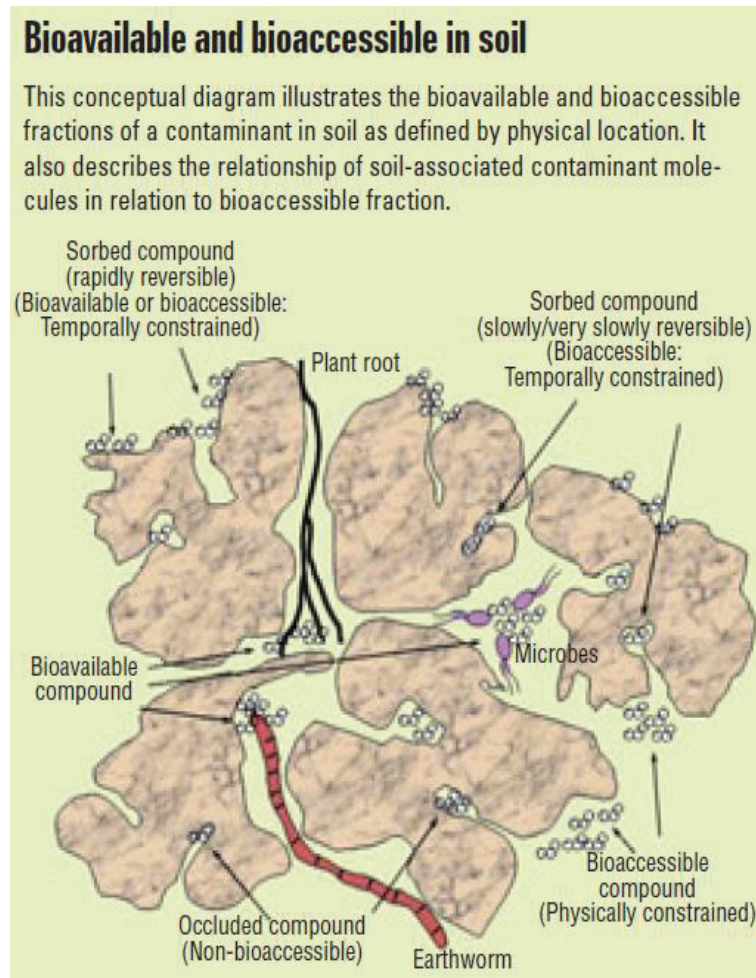


Figure 11: Biodisponibilité et bioaccessibilité des contaminants dans les sols (Semple *et al.*, 2004).

Aujourd'hui, nous ne sommes pas capables de mesurer la concentration totale d'un nanomatériau dans un sol en raison de limitations techniques. Évaluer la fraction biodisponible et bioaccessible des nanomatériaux est un défi encore plus grand qui nécessite des développements méthodologiques importants et donc les données à ce sujet sont quasi-inexistantes (Vittori Antisari *et al.*, 2013). Toutefois, lorsque la toxicité des nanomatériaux repose sur la dissolution de métaux comme dans le cas de l'Ag ou du Zn, les études peuvent se baser sur des méthodes bien connues pour évaluer la fraction biodisponible des éléments

traces dissous (Cornelis *et al.*, 2014). Même si nous ne pouvons pas mesurer la fraction biodisponible, de nombreuses études ont montré que les nanomatériaux étaient retrouvés dans les racines ou les feuilles de différentes plantes (Rico *et al.*, 2011), dans les tissus d'invertébrés (Tourinho *et al.*, 2012) ou dans des champignons mycorhiziens (Whiteside *et al.*, 2009), illustrant leur biodisponibilité réelle dans les sols. Toutefois, il faut rester prudent, un composé biodisponible n'est pas nécessairement toxique, selon la concentration présente et les propriétés du polluant dans l'environnement considéré.

b. Toxicité des nanomatériaux dans les sols

De nombreuses études *in vitro* ont montré la toxicité des nanomatériaux et identifié des mécanismes de toxicité (e.g. Nel *et al.* 2006). Les principaux mécanismes invoqués reposent principalement sur l'adsorption de NPs à la surface des cellules, leur dissolution et des stress oxydatifs induits par la production de ROS (H_2O_2 , $OH\cdot$, $O_2\cdot^-$, 1O_2) (Figure 12 ; (Nel *et al.*, 2006; Handy *et al.*, 2008; Jiang *et al.*, 2008; Simon-Deckers *et al.*, 2009; Li *et al.*, 2012). De nombreux effets au niveau des membranes bactériennes ont été rapportés, comme la dépolarisation ou l'augmentation de la perméabilité ainsi que des dommages au niveau de protéines ou de l'ADN ((Jomini *et al.*, 2012; Sohm *et al.*, 2015; Bondarenko *et al.*, 2012).

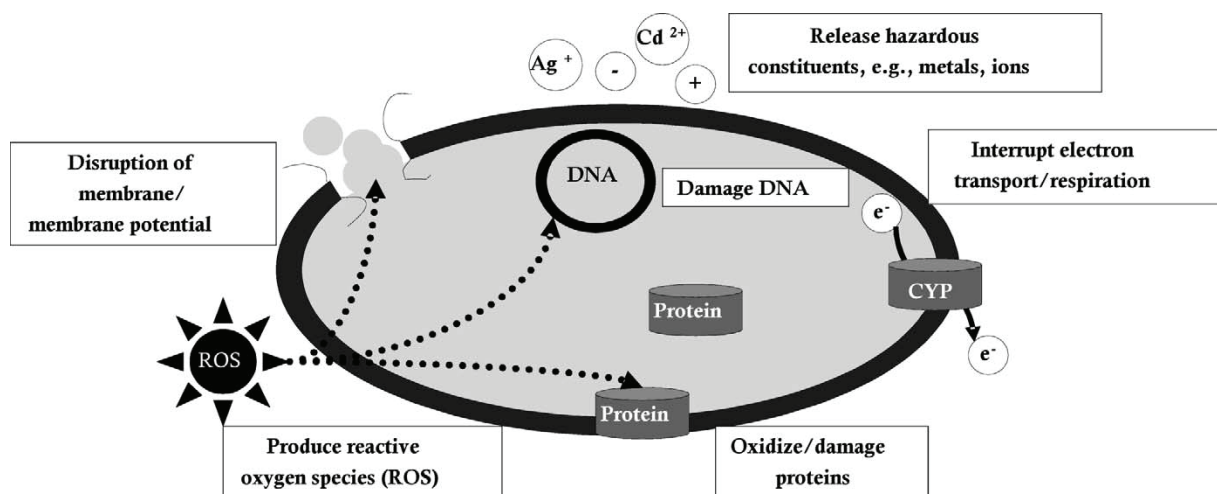


Figure 12: Schématisation des possibles mécanismes de toxicité des nanomatériaux sur des cellules bactériennes. CYP=cytochrome P (Handy *et al.*, 2008)

Toutefois, il est difficile de savoir si ces mécanismes de toxicité observés *in vitro* seront similaires en conditions environnementales. En effet, à partir des connaissances disponibles sur les transformations des nanomatériaux dans les sols (Partie 2. a.), on s'attend à ce que

les facteurs abiotiques influencent les propriétés physico-chimiques et donc la toxicité de ces contaminants. Un paramètre sans doute clé pour leur toxicité est l'agrégation, car elle réduit l'aire de surface « disponible » et donc leur réactivité (Hotze *et al.*, 2010). L'agrégation pourrait donc réduire leur toxicité lorsqu'elle est associée à des processus de surface, comme l'émission de ROS et la dissolution (Lowry *et al.*, 2012). D'autre part, l'hétéroagrégation entraîne une augmentation de la taille effective des nanomatériaux, ce qui pourrait limiter leur ingestion par certains organismes ou le passage à travers une paroi et/ou membrane cellulaire (Tourinho *et al.*, 2012). Ainsi certains paramètres physico-chimiques du sol identifiés pour favoriser ou limiter l'hétéroagrégation pourraient être de bons indicateurs de la biodisponibilité des nanomatériaux. Une force ionique élevée et la présence d'argile semblent être des conditions favorables à une forte hétéroagrégation des NPs et conduirait donc à de faibles biodisponibilité et toxicité (Shoults-Wilson *et al.*, 2011). Cependant en présence de MO dissoute, il a été observé une augmentation de l'écotoxicité/accumulation associée à une augmentation de la mobilité dans les sols (Cornelis *et al.*, 2014). De ce fait l'hétéroagrégation n'implique pas forcément une diminution de la biodisponibilité et de la toxicité des nanomatériaux dans les sols.

L'évaluation de la biodisponibilité et de la toxicité des nanomatériaux et des facteurs du sol les influençant en est encore à ses balbutiements, mais quelques directions à suivre pour les études futures peuvent d'ores et déjà être identifiées. La première est l'amélioration des méthodologies pour caractériser les nanomatériaux dans des conditions environnementales proches du sol et le développement des protocoles permettant d'évaluer la fraction biodisponible/bioaccessible des NPs en milieux hétérogènes complexes. Le second point est l'évaluation de l'influence des propriétés du sol, telles que la texture (teneur en argile, limon, sable), de la force ionique et de la teneur en MO sur l'écotoxicité des nanomatériaux, car il semble probable que ces facteurs affecteront leur biodisponibilité et toxicité.

4. Impact des nanomatériaux sur les communautés microbiennes du sol

L'écotoxicité des nanomatériaux dans les sols peut être évaluée en s'intéressant à leur impact sur différents organismes et/ou communautés d'organismes retrouvés dans les sols (e.g. microorganismes, vers de terre, nématodes). Parmi eux, les communautés microbiennes, à savoir les bactéries, les archées et les champignons, sont reconnues comme étant des indicateurs pertinents et sensibles pour l'évaluation de l'impact des polluants sur le fonctionnement biologique du sol (Kandeler *et al.*, 1996; Giller *et al.*, 1998; Schloter *et al.*, 2003; Holden *et al.*, 2014). En effet, du fait de leur rôle dans les cycles biogéochimiques, les communautés microbiennes sont impliquées dans des services écosystémiques associés à la fertilité et la production végétale. Les microorganismes ont également un rôle clé dans la dégradation des polluants dans les sols (Leahy and Colwell, 1990).

La réponse des communautés microbiennes à des perturbations environnementales et notamment aux stress chimiques, peut être appréhendée en évaluant de façon complémentaire les effets sur l'activité, l'abondance et la diversité des communautés microbiennes (Griffiths and Philippot, 2013). Une synthèse bibliographique dédiée spécifiquement aux effets des NPs sur les communautés microbiennes de sol a été rédigée.

a. « Impact of engineered nanoparticles on the activity, abundance, and diversity of soil microbial communities: a review »

L'article 1 intitulé « Impact of engineered nanoparticles on the activity, abundance, and diversity of soil microbial communities: a review » a été publié en 2015 dans le Journal *Environmental Science and Pollution Research* dans le cadre d'un numéro spécial *Microbial Ecology of the Terrestrial and Coastal Environments*.

Impact of engineered nanoparticles on the activity, abundance, and diversity of soil microbial communities: a review

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Abstract This report presents an exhaustive literature review of the effects of engineered nanoparticles on soil microbial communities. The toxic effects on microbial communities are highly dependent on the type of nanoparticles considered. Inorganic nanoparticles (metal and metal oxide) seem to have a greater toxic potential than organic nanoparticles (fullerenes and carbon nanotubes) on soil microorganisms. Detrimental effects of metal and metal oxide nanoparticles on microbial activity, abundance, and diversity have been demonstrated, even for very low concentrations ($<1 \text{ mg kg}^{-1}$). On the opposite, the negative effects of carbon nanoparticles are observed only in presence of high concentrations ($>250 \text{ mg kg}^{-1}$), representing a worst case scenario. Considering that most of the available literature has analyzed the impact of an acute contamination of nanoparticles using high concentrations in a single soil, several research needs have been identified, and new directions have been proposed. The effects of realistic concentrations of nanoparticles based on the concentrations predicted in modelization studies and chronic contaminations should be simulated. The influence of soil properties on the nanoparticle toxicity is still unknown and that is why it is crucial to consider the ecotoxicity of nanoparticles in a range

of different soils. The identification of soil parameters controlling the bioavailability and toxicity of nanoparticles is fundamental for a better environmental risk assessment.

Keywords Nanomaterials · Microbial ecotoxicology · Terrestrial ecosystem · Soil pollution · Risk assessment · Nanoscale zero valent iron

Introduction

Because of their unique properties, engineered nanoparticles (NPs) are increasingly produced and used in diverse commercial products (agriculture, cosmetics, energy, electronics, paint, medicine...). Toxic effects of NPs have been observed in numerous *in vitro* studies; in particular, many NPs, such as silver-, copper-, or zinc oxides-NPs, are known to have antimicrobial properties (Neal 2008; Fabrega et al. 2009; Dinesh et al. 2012). The toxicity mechanism often mentioned is an oxidative stress generated by the production of reactive oxygen species (ROS) from NPs in contact with microbial membranes, causing disruption of membranes, oxidation of proteins, or interruption of energy transduction (Klaine et al. 2008; Neal 2008; Xia et al. 2008).

During the different phases of their life cycle, from production to disposal, NPs can be released to the environment, raising great concerns about potential ecological risks. Direct measurement of NP concentration in the environment remains difficult due to current technical limitations (Cornelis et al. 2014). Currently, the only way to get information on existing levels of NPs in the environment is to model predicted environmental concentrations (Sun et al. 2014). Exposure modeling strongly suggests that soil could be a major sink of NPs

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compared to air and water ecosystems (Gottschalk et al. 2009; Keller et al. 2013; Sun et al. 2014).

NPs can enter soil through various pathways, such as agricultural amendments of sewage sludge, atmospheric deposition, landfills, or accidental spills during industrial production. The current models estimate that in sewage sludge-treated soil, TiO₂-NP concentrations increase between 0.94 and 3.6 mg kg⁻¹ per year, whereas for Ag-NPs and fullerenes, the yearly predicted increases are more than 1000-fold lower. In this case, concentration increases are between 0.09 and 0.65 µg kg⁻¹ and between 0.38 and 1.5 µg kg⁻¹, respectively (Sun et al. 2014). Intentional applications of NPs can also be possible in a context of soil remediation. Nanoscale zero-valent iron (nZVI) particles have the potential to remediate diverse environmental contaminants such as chlorinated organic compounds or inorganic compounds for example. This remediation strategy has been mainly employed for the decontamination of groundwater, and its utility in soil remediation is being increasingly considered (Naja et al. 2009; Satapanajaru et al. 2008). Therefore, there is a demand to assess the risks associated to NPs in soils, in order to preserve the soil capacity to fulfill essential ecosystem services.

Soil microbial communities are both relevant and sensitive indicators of soil perturbations (Brookes 1995; Kandeler et al. 1996; Holden et al. 2014) because of their key role in biogeochemical cycling (carbon, nitrogen, phosphorous, sulfur cycles), biodegradation of pollutants, crop production, and climate. A microbial ecotoxicological approach based on the response of soil microbial communities offers the opportunity to evaluate the impact of pollutants on natural assemblages of populations, contrary to toxicological studies which use single populations in synthetic media. This approach allows a realistic assessment of the response of natural communities to NP contamination and leads consequently to a better environmental risk assessment. Environmental constraints and/or anthropogenic perturbation can trigger a microbial response observable through different variables among the community. The microbial community responses to a disturbance can be monitored using (i) activity rates, (ii) abundances, and (iii) diversity (Brookes 1995; Griffiths and Philippot 2013). Through this review, we provide a synthesis of studies assessing the overall impact of NPs on soil biodiversity and functioning using the descriptors mentioned above.

Among NPs, a wide range of different materials with diverse physical, chemical, and toxicological properties exist. Inorganic and organic NPs can be distinguished based on their core material (Ju-Nam and Lead 2008). Inorganic NPs are divided into metal, metal oxide, and quantum dots NPs, while fullerenes and carbon nanotubes (single and multi-walled CNTs) are defined as organic NPs. Since NPs cannot be considered as a single homogeneous group of contaminants, we will review separately the effects of (i) metal and metal oxides, (ii) carbon nanoparticles, and (iii) nZVI. Due to increasing

interest of this latter for in situ soil remediation, a specific attention will be addressed to the nZVI. The direct application to soils of large amounts of nZVI for remediation purposes raise specific concerns about potential consequences on soil microbial communities and their key functions for soil fertility and biodegradation of pollutants.

Impact of nanoparticles on microbial activities

Soil microbial activities are good indicators of soil quality, since soil microorganisms control the transformation and mineralization of natural compounds and xenobiotics (Schloter et al. 2003). The impact of perturbations on microbial activity is classically approached through measurements of “broad” processes (Schimel and Schaeffer 2012) of soil respiration or generalist enzyme activities such as β-glucosidase, urease, and dehydrogenase. Some specific transformations or “narrow” processes realized by a phylogenetically constrained group of microorganisms (Schimel and Schaeffer 2012) can be also analyzed, such as nitrification in nitrogen cycle or the ability to degrade specific pollutants.

Impact of metal and metal oxide nanoparticles

The use of microbial activities to assess the effect of metal or metal oxide NPs is often encountered in the literature (Table 1). Contrasted responses can be observed according to the type of NPs, the concentration, the exposure time, and the measured activity.

Ag-NP has been found to reduce substrate-induced respiration and enzymatic activities after 50 days of exposure to a low concentration of Ag-NPs (0.14 mg kg⁻¹ soil) applied in mesocosms via sewage sludge (Colman et al. 2013). Another study using much lower concentrations of Ag-NPs (0.0032, 0.032, 0.32 mg kg⁻¹ soil) did not show any changes in the different tested enzymatic activities (Hänsch and Emmerling 2010). However, the authors have found a reduction of the net nitrogen mineralization and an increase of the metabolic quotient suggesting a reduction in substrate use efficiency in presence of Ag-NPs. Shin et al. (2012) have also reported adverse effects of Ag-NPs on enzymatic activities, especially on urease, but the larger reductions were observed only for high concentrations (100 and 1000 mg kg⁻¹ soil). Using soluble silver ions as a positive control, these studies show that NPs themselves were responsible for the negative effects on soil microorganisms but not the silver ions dissolved from Ag-NPs (Shin et al. 2012; Colman et al. 2013).

Fe₂O₃-NPs stimulated urease and invertase activities, whereas Fe₃O₄-NPs did not induce any modification of enzymatic activities in presence of high concentrations (420, 840, and 1260 mg kg⁻¹ soil) (He et al. 2011). These stimulations

Table 1 Review of the effects of metal and metal oxide nanoparticles on soil microbial communities

Metal and metal oxide nanoparticles		Metal and metal oxide nanoparticles		Metal and metal oxide nanoparticles		Metal and metal oxide nanoparticles	
Reference	NPs	NPs concentrations in soil	Soil	Experiment	Major results		
Colman et al. (2013)	Ag-NP	0.14 mg kg ⁻¹	Sandy loam (USA)	Mesocosms field experiment- Application of sewage sludge. 0,1 and 50 days	Modification of bacterial community structure (T-RELP) after 1 day. Stimulation of N ₂ O fluxes after 8 days. Reduction of SIR and enzymatical activities (phosphatase and leucine aminopeptidase) after 50 days. Larger effect of Ag-NP than the positive control AgNO ₃ .		
Hänsch and Emmerling (2010)	Ag-NP	3.2, 32, 320 µg kg ⁻¹	Sandy-loam (Germany)	Mesocosms-1, 7,14 and 120 days	Decrease of microbial biomass in a dose-response manner. Increases of basal respiration and metabolic quotients. No effect on microbial biomass N and fluorimetric enzymes.		
Peyrot et al. (2014)	Ag-NP	1.25 µg to 30 mg kg ⁻¹	Sandy soil with or without compost amendment (Canada)	Microcosms - 6 weeks	Decrease of enzymatic activities (phosphomonoesterase, β-D-glucosidase, arylsulfatase, leucine-aminopeptidase), especially at the low Ag concentrations. No protective role of OM was observed because concentrations of dissolved-Ag were similar in both amended and unamended soils.		
Shan et al. (2012)	Citrate coated Ag-NP	1, 10, 100 and 1000 mg kg ⁻¹	Sandy (Korea)	Microcosms - 1 and 7 days	Strong negative dose-response effect on urease activity. Reduction of 5 enzymatic activities in the two highest concentrations (acid phosphatase, arylsulfatase, β-glucosidase, dehydrogenase and fluorescem diacetate hydrolase). The effects of silver ions dissolved from the Ag-NP were not significant, indicating the adverse effects caused by Ag-NP themselves.		
Vitort Antisari et al. (2013)	CeO ₂ -NP, Fe ₃ O ₄ -NP, SnO ₂ -NP	0, 10 and 100 mg kg ⁻¹	Sandy clay loam (Italy)	Microcosms - 7 and 60 days	No effect on C and N microbial biomass. Increased of metabolic quotient (qCO ₂) in the polluted soil.		
Kumar et al. (2011)	Cu-NP, Ag-NP, Si-NP	660 mg kg ⁻¹	Sandy -peaty Arctic soil (Canada)	Microcosms - 176 days	Reduction of the diversity of substrate utilization (Biolog) to 16 % in Ag NP-treated soil, and to 40 % and 52 % in Cu and SiO ₂ NP-treated soil. No effect on FAME profiles. Ag-NP caused a modification of bacterial composition (DGGE analysis).		
Kumar et al. (2012)	Mixture of Cu-NP, Ag-NP and Si-NP	220 mg kg ⁻¹ soil x 3 NPs = 660 mg kg ⁻¹	Sandy -peaty Arctic soil (Canada)	Microcosms - 176 days	Reduction of C and N microbial biomass. Modification of microbial community structure (FAME analysis) and especially bacterial community (DGGE analysis). No effect on microbial community level physiological profiles.		
Ben-Moshe et al. (2013)	CuO, Fe ₃ O ₄	10 g kg ⁻¹	2 soils: a red sandy clay loam soil and a Rendzina soil (Israel)	Microcosms - 24 or 48 h incubation	NPs affected the soil bacterial community composition, based on DGGE fingerprinting. Both soil types were less affected by Fe ₃ O ₄ contamination than by CuO contamination, suggesting lower toxicity of Fe ₃ O ₄ to soil bacterial communities.		
Rousk et al. (2012)	CuO-NP, ZnO-NP	8 concentrations at logarithmic intervals from 0 to 200 nmol metal g ⁻¹	2 soils: a «mineral» and an «organic» soil (sandy loam) (UK)	Microcosms - 5-7 h of incubation	CuO-NP decreased bacterial growth (leucine incorporation method) only in the «mineral» soil. NP ZnO reduced bacterial growth in both soil. The toxic effect of these NP may be explained by their dissolved forms.		
Collins et al. (2012)	Cu ⁰ -NP, ZnO-NP	Application of 550 mg of NP powder at the center of mesocosms	Agricultural soil (USA)	Field mesocosms- 1, 7, 30 and 162 days	Effect on microbial community level physiological profiles only after 1 day. No effect on FAME profiles. Potential alteration of bacterial community structure (pyrosequencing 454), especially Flavobacteriales and Sphingomonadales.		
He et al. (2011)	Fe ₃ O ₄ -NP, γ-Fe ₂ O ₃ -NP	420, 840 and 1260 mg kg ⁻¹	Agricultural soil (China)	Microcosms - 0, 15 and 30 days	No effect on bacterial abundance. Increase of enzymatic activities (urease and invertase) that could be explained by a stimulation of specific groups of Acinobacteria (DGGE analysis).		
Ce et al. (2011)	TiO ₂ -NP, ZnO-NP	For TiO ₂ : 500, 1000 and 2000 mg kg ⁻¹ soil For ZnO: 50, 100 and 500 mg kg ⁻¹ soil	Loam (USA)	Microcosms - 15 and 60 days	Decreases of SIR and microbial biomass (DNA quantity) by both NPs, with different dose-response effects: linear for TiO ₂ -NP and exponential for ZnO-NP. No effect on basal respiration. TiO ₂ -NP		

Table 1 (continued)

Metal and metal oxide nanoparticles	
Ge et al. (2012)	<p>TiO₂-NP, ZnO-NP</p> <p>For TiO₂: 500, 1000 and 2000 mg kg⁻¹ soil For ZnO: 50, 100 and 500 mg kg⁻¹ soil</p> <p>Loam (USA)</p> <p>Microcosms - 15 and 60 days</p> <p>and ZnO-NP altered soil bacterial community structure and decreased bacterial diversity (T-RFLP analysis). (Pyrosequencing 454) associated to a decrease of 12 taxa: class <i>Alphaproteobacteria</i>, the order <i>Rhizobiales</i>, families <i>Bradyrhizobiaceae</i>, <i>Geodermatophilaceae</i>, <i>Methylobacteriaceae</i>, <i>Micromonosporaceae</i>, and <i>Rhodospirillaceae</i>, and the genera <i>Actinoplanes</i>, <i>Rubrinatus</i>, <i>Blastococcus</i>, <i>Bradyrhizobium</i>, and <i>Sternumella</i>. Increase of the abundance of 3 taxa (<i>Sphingomonadaceae</i>, <i>Streptomycetaceae</i> and <i>Streptomyces</i>) known to be associated with the decomposition of recalcitrant organic pollutants and biopolymers. The abundance of these 3 taxa was correlated to an increase of protease activity.</p>
Ge et al. (2013)	<p>TiO₂-NP</p> <p>20 g kg⁻¹</p> <p>Sandy-clay-loam (USA)</p> <p>Microcosms-288 days</p> <p>Decrease of bacterial diversity and modification of bacterial community structure (T-RFLP analysis).</p>
Simoin et al. (2014)	<p>TiO₂-NP</p> <p>1 and 500 mg kg⁻¹</p> <p>6 soils: sandy-loam, loam and silty-clay with low or high OM content (France)</p> <p>Microcosms - 7, 30 and 90 days</p> <p>Decrease of substrate-induced respiration only in one soil (silty-clay soil with high OM content), even for the low concentration. No effect on bacterial abundance.</p>
Du et al. (2011)	<p>TiO₂-NP, ZnO-NP</p> <p>TiO₂-NP: 91 mg kg⁻¹ soil ZnO-NP: 45 mg kg⁻¹ soil</p> <p>Silty - clay (China)</p> <p>Field lysimeters - 9 months</p> <p>TiO₂-NP: Increase of urease and decreases of catalase and peroxidase. ZnO-NP: Decreases of protease, catalase and peroxidase.</p>
Nogueira et al. (2012)	<p>TiO₂-NP, TiSiO₂-NP, CuSe/ZnS quantum dots, gold nanorods, Fe/Co magnetic fluid</p> <p>TiO₂-NP and TiSiO₂-NP: 5 g kg⁻¹ soil CuSe/ZnS quantum dots and Fe/Co magnetic fluid: 0.5 mg kg⁻¹ soil gold nanorods: 3.34 mg kg⁻¹ soil</p> <p>Standard artificial OECD soil</p> <p>Microcosms - 30 days</p> <p>Higher impact of TiO₂ and gold nanorods in the structural diversity of bacterial community (DGGE analysis). No effect or limited effects of TiSiO₂, CuSe/ZnS quantum dots and Fe/Co magnetic fluid NPs on DGGE profiles.</p>
Shah and Belozerova (2009)	<p>Si-NP, Pd-NP, Au-NP, Cu-NP</p> <p>130 and 660 mg kg⁻¹ soil</p> <p>Soil (USA)+potting mix</p> <p>Microcosms- 15 days</p> <p>No effect of the NPs on substrate utilization patterns of microbial communities, nor on FAME profiles.</p>
Shah et al. (2014)	<p>Fe-NP, Co-NP, Ni-NP</p> <p>Application of 550 mg of NP powder at the center of mesocosms</p> <p>Field mesocosms - 50 days</p> <p>No decrease of bacterial richness (Pyrosequencing 454) but some genera were affected by the NPs. <i>Sphingomonas</i> and <i>Lysobacter</i> genera were decreased, whereas <i>Flavobacterium</i> and <i>Mastella</i> genera were increased.</p>

were explained as a consequence of changes observed within the bacterial community.

Aged TiO₂-NPs (91 mg kg⁻¹ soil) stimulated urease activity but decreased catalase and peroxidase activities after 9 months of incubation in lysimeters (Du et al. 2011). In the same experiment, ZnO-NPs (45 mg kg⁻¹ soil) reduced protease, catalase, and peroxidase activities. Ge et al. (2011) also evaluated the impact of TiO₂ and ZnO-NPs on substrate-induced respiration and found a reduction after 15 and 60 days with 500, 1000, and 2000 mg kg⁻¹ soil. TiO₂-NPs impact on substrate-induced respiration was also assessed in six contrasted agricultural soils (1 and 500 mg kg⁻¹), and a significant decrease of this activity was noted only in one soil presenting a silty clay texture and a high OM content (Simonin et al. 2014).

These results indicate that metal and metal oxides can induce modifications of microbial activities in soil and consequently on biogeochemical cycles. The decrease of respiration and enzymatic activities in response to very low concentrations of Ag-NPs illustrates the high toxic potential of some metal NPs.

Impact of carbon nanoparticles (fullerene and CNTs)

Fullerene (C₆₀) had no adverse effect on soil respiration or enzymatic activities (Table 1, Tong et al. 2007; Johansen et al. 2008) even after 180 days of incubation. In the same range of concentrations, no modification of microbial activities (soil respiration and enzymatic activities) was reported for MWCNT (Shrestha et al. 2013), except for the highest concentrations used (500 and 5000 mg kg⁻¹ soil) (Chung et al. 2011). Likewise, SWCNT caused a reduction of the enzymatic activities only at high concentrations (300 to 1000 mg kg⁻¹ soil) (Jin et al. 2013).

Taken together, these data suggest that carbon NPs have a very low toxic potential to soil microorganisms compared to metal and metal oxide NPs, because modifications of microbial activities were only observed at very high concentrations (>500 mg kg⁻¹ soil) representing the worst case scenario given the present annual production of CNTs (Keller et al. 2013).

Impact of nanoscale zero valent iron

Tilston et al. (2013) performed an experiment with a sandy loam soil artificially contaminated with the PCB Aroclor-1242 to simulate the utilization of nZVI in a context of remediation of organochlorine-contaminated environments. The authors reported a decrease of the activity of chloroaromatic mineralizing microorganisms (2,4-D mineralization), which biodegradative functions could contribute to contaminant remediation. Nevertheless, this negative effect should be taken with caution due to the possible induction of confounding factors as shown by Cullen et al. (2011). No conclusive

evidence for negative effects of nZVI was observed due to the redox properties of these particles that induced a significant abiotic modification of the concentration of either the product or substrate of assays for nitrification or deshydrogenase activities. When examining the impact of redox active particles such as nZVI on microbial oxidation–reduction reactions, sterile controls should be made to take into consideration these potential confounding factors (Cullen et al. 2011).

The influence of environmental physicochemical properties on nZVI toxicity and bioavailability was investigated by comparing their impact in different soils. Pawlett et al. (2013) found that microbial respiration induced by multiple substrates was decreased in a clay soil spiked with 270 mg nZVI kg⁻¹ soil but not in sandy or loam soils for the same dose.

Impact of nanoparticles on microbial biomass and abundance of functional groups

Microbial biomass is a sensitive indicator of pollution in soils (Brookes 1995). It has been used as an indicator of the impact of NPs in the majority of studies assessing their ecotoxicity. Different methodologies are used in NPs ecotoxicology studies to quantify microbial biomass in soils, such as fumigation–extraction or extractable soil DNA (Hänsch and Emmerling 2010; Ge et al. 2011). In addition to these approaches that assess the total biomass including bacteria, fungi, and archaea, the abundance of specific groups of microorganism targeting universal genes (*16S* or *18SrRNA* genes) or functional genes can be assessed using quantitative PCR (Fajardo et al. 2012).

Impact of metal and metal oxide nanoparticles

Different concentrations of Ag-NPs (0.0032, 0.032, 0.32 mg kg⁻¹ soil) induced a decrease in microbial biomass in a dose–response manner (Hänsch and Emmerling 2010). However, soil contamination with iron oxide magnetic NPs (Fe₂O₃ and Fe₃O₄) had no effect on microbial biomass with 10 and 100 mg kg⁻¹ soil (Vittori Antisari et al. 2013) nor on bacterial abundance with 420, 840, and 1260 mg kg⁻¹ soil (He et al. 2011). Similarly, Simonin et al. (2014) found no significant effect of TiO₂-NPs on bacterial abundances in the six soils tested. TiO₂ and ZnO NPs using 500, 1000, and 2000 mg kg⁻¹ soil, both reduced extractable soil DNA (Ge et al. 2011). However, the dose–response curves were linear with TiO₂-NPs and exponential with ZnO-NPs (Ge et al. 2011). These differences were explained by contrasting bioavailability and environmental behaviors of these metal oxide NPs.

The simultaneous environmental exposure of soils to different NPs is likely to occur for example after sewage sludge application. The impact of a combination of NPs has been

investigated using Ag, Cu, and Si NPs in an arctic soil (Kumar et al. 2011). In this study, a significant decrease of microbial biomass was observed after 6 months of incubation for a mixture concentration of 660 mg kg⁻¹ soil (220 mg kg⁻¹ of each NP).

Impact of carbon nanoparticles (fullerene and CNTs)

Contrasting effects on microbial biomass were reported for carbon nanoparticles. Johansen et al. (2008) found no effect, whereas decreases were observed by other authors but only for concentrations exceeding 250 mg kg⁻¹ of CNTs (Chung et al. 2011; Jin et al. 2013; Rodrigues et al. 2013). These negative effects seemed to be more severe on fungal biomass compared to bacterial biomass (Rodrigues et al. 2013). These effects observed only with high concentrations are consistent with the results previously presented on the microbial activities, suggesting a limited toxicity of carbon NPs toward soil microbial communities.

Impact of nanoscale zero valent iron

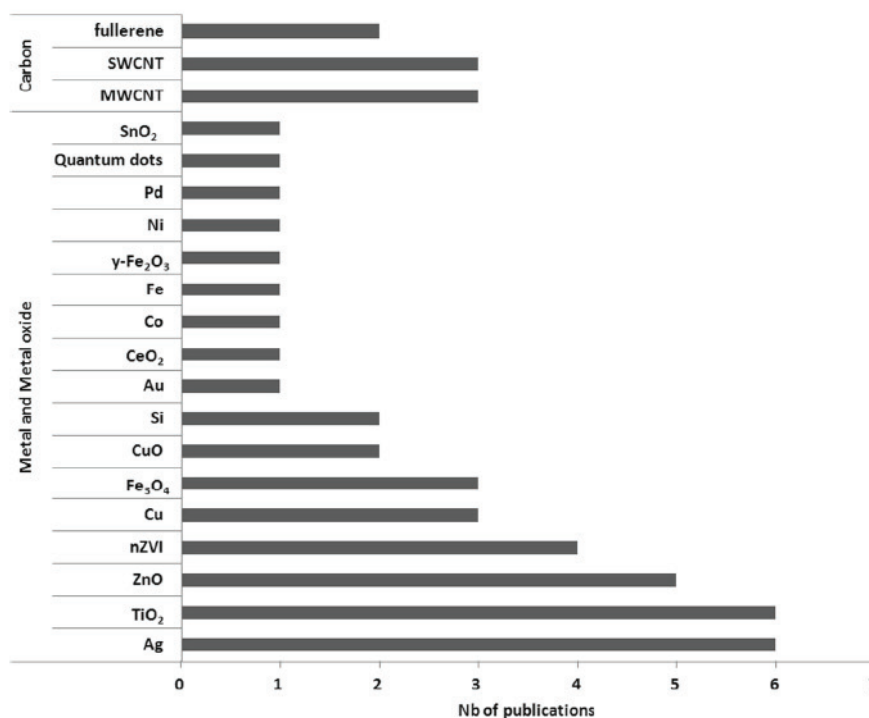
The effects of nZVI have been investigated on the abundance of microbial functional groups to detect potential modifications that could be detrimental for ecosystem functioning. The presence of 34 and 10 g nZVI kg⁻¹ soil triggered decrease in denitrifying bacteria abundance (Fajardo et al. 2012) and in

chloroaromatic mineralizing microorganisms (Tilston et al. 2013), respectively. These results illustrate that a high concentration of nZVI could affect soil nitrogen cycle and the biodegradative potential of a microbial functional group. Pawlett et al. (2013) reported that nZVI addition reduced microbial biomass but only when soil was amended with 5 % straw. These results suggest that the impact of NPs may be dependent of the organic matter content of soil, which has been found to enhance NPs mobility in porous media (Ben-Moshe et al. 2010; Wang et al. 2012; Thio et al. 2011). Organic matter may be an important factor favoring NPs bioavailability for soil microorganisms. However, the number of studies available on the mobility and toxicity of NPs in natural soils with contrasted textures and organic matter contents is too limited to confirm this assumption (Fang et al. 2009; Pawlett et al. 2013; Cornelis et al. 2013; Peyrot et al. 2014; Simonin et al. 2014).

Impact of nanoparticles on microbial diversity

The measurement of the microbial biomass is a black box approach in which the genetic and functional diversity among the microbial community are not considered. Soil hosts an immense diversity of microorganisms (individual taxa commonly described as “operational taxonomic units”; OTUs) of bacteria, fungi, and archaea. Microbial diversity encompasses

Fig. 1 Number of publications studying the impact of metal/metal oxide or carbon nanoparticles on soil microbial communities. Thirty-one publications were available in July 2014



genetic variability within taxa (species), and the number (richness) and the relative abundance (evenness) of taxa and functional groups (guilds) in communities (Torsvik and Øvreås 2002). That is why, investigating the impact of NPs on microbial diversity is crucial to provide information on how and why soil ecosystem functioning is affected. A panel of techniques, such as PLFA profiles, PCR-DGGE, T-RFLP analysis, or next generation sequencing, has been used to evaluate the impact of NPs on microbial diversity.

Impact of metal and metal oxide nanoparticles

Ag-NPs altered bacterial community structure, after short-term exposure of sewage sludge containing 0.14 mg kg⁻¹ of Ag-NP (Colman et al. 2013). Kumar et al. (2011) also observed that Ag-NPs could modify bacterial community in an arctic soil but with a higher concentration (660 mg kg⁻¹). A field study also reported that zero valent Cu and ZnO-NPs had no effect on PLFA profiles and on microbial community composition as determined by pyrosequencing (Collins et al. 2012). Changes in soil bacterial community composition due to the presence of Fe₃O₄-NPs were observed after a short

incubation (24 or 48 h) as well as after 15 and 30 days (Ben-Moshe et al. 2013; He et al. 2011). Cloning–sequencing of DGGE bands indicated a stimulation of specific groups of *Actinobacteria*, *Duganella*, *Streptomyetaceae*, or *Nocardioidea* (He et al. 2011). These groups facilitate the decomposition of organic matter, which could explain the concomitant soil invertase and urease increases measured during this experiment. Different concentrations of TiO₂ and ZnO-NPs decreased soil bacterial diversity after 60 days of incubation (Ge et al. 2011; Ge et al. 2012). The pyrosequencing data indicated that some of the declining taxa are known to be associated to nitrogen fixation (*Rhizobiales*, *Bradyrhizobiaceae*, and *Bradyrhizobium*) and methane oxidation (*Methylobacteriaceae*), while some positively impacted taxa are known to be associated to the decomposition of recalcitrant organic pollutants (*Sphingomonadaceae*) and biopolymers including protein (*Streptomyetaceae* and *Streptomyces*). The role of these taxa for soil functioning suggest potential consequences on ecosystem-scale processes. Nogueira et al. (2012) assessed the effect of five inorganic nanomaterials (TiO₂, TiSiO₄, CdSe/ZnS quantum dots, gold nanorods, and Fe/Co magnetic fluid) on soil bacterial

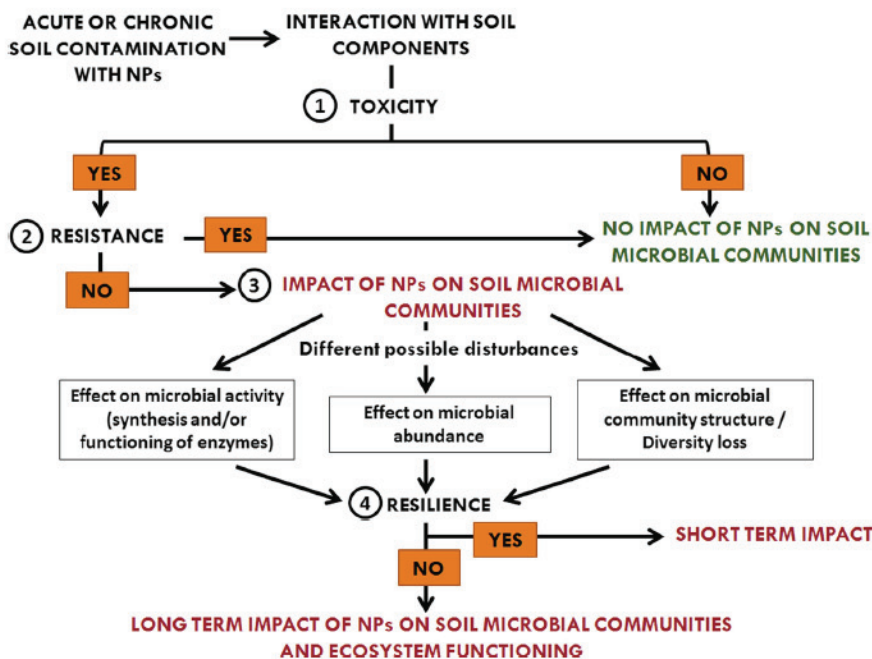


Fig. 2 The response of soil microbial communities to NPs contamination. 1 Soil properties (clay, organic matter content, pH, ionic strength...) can modulate the toxicity of NPs for microorganisms. 2 If the NPs can be toxic, the impact on microbial community will depend on the ability of the community to resist to this perturbation. 3 If the indigenous microorganisms are not resistant to NPs contamination, soil microbial communities could be affected through different nonexclusive mechanisms: (i) effects of NPs on microbial activities due to an alteration of the synthesis and/or functioning of enzymes, (ii) a

modification of microbial abundance caused by a mortality of sensitive population in presence of NPs, and (iii) a modification of microbial community structure and/or a diversity loss. 4 Soil microbial communities may become tolerant to NPs contamination and may be resilient to this disturbance. If resilience is observed, the impact of NPs will be on relatively short-term duration. If no or limited resilience is observed, NPs could have long-lasting effects on soil microbial communities with cascading effects on ecosystem functioning, especially biogeochemical cycles, soil fertility, and climate regulation

Table 2 Review of the effects of carbon nanoparticles on soil microbial communities

Carbon nanoparticles (fullerene and CNTs)					
Reference	NPs	NPs concentrations	Soil	Experiment	Major results
Tong et al. (2007)	FullereneC ₆₀	1 mg kg ⁻¹ soil in aqueous suspension or 1000 mg kg ⁻¹ soil in granular form	Silty-clay-loam (USA)	Microcosms- up to 180 days	No effect of fullerene on soil respiration, enzymatic activities PLFA profiles or DGGE profiles.
Johansen et al. (2008)	FullereneC ₆₀	0, 5, 25, 50 mg kg ⁻¹ soil	Sandy-clay-loam (Denmark)	Microcosms - 0, 7, 14 days	No effect of fullerene on soil respiration, microbial biomass, and on the enumeration of protozoans. Decrease of bacterial enumeration only immediately after contamination. Limited effects of fullerene on DGGE profiles (15-30 % dissimilarity between treatments) for both Bacteria and Protozoans.
Chung et al. (2011)	MWCNT	50, 500, 5000 mg kg ⁻¹ soil	2 sandy soils (Korea)	Microcosms - 0, 1, 4, 11 days	In both soil types, most enzyme activities decreased at 500 mg kg ⁻¹ soil, and all enzymatic activities as well as microbial biomass C and N were significantly lowered at 5000 mg kg ⁻¹ soil.
Khodakovskaya et al. (2013)	MWCNT	50 or 200 µg mL ⁻¹ added once a week	Redi-earth Plug and Seedling Mix (Sun Gro horticulture, Inc.)	Experimental pot seeded with tomato - 9 weeks	No decrease of bacterial diversity was observed but a significant modification of the bacterial composition was found (DGGE analysis). The relative abundances of Bacteroidetes and Firmicutes increased, whereas Proteobacteria and Verrucomicrobia decreased with increasing concentration of MWCNT.
Shrestha et al. (2013)	MWCNT	10, 100, 1000 mg kg ⁻¹ soil	Sandy loam (USA)	Jars - 28 and 90 days	No effects of MWNTs on soil respiration, enzymatic activities, and microbial community composition. Decreased abundance of some bacterial genera like Derrxia, Holophaga, Opitutus and Waddlia at the highest treatment while bacterial genera that are considered potential degraders of recalcitrant contaminants (such as polycyclic aromatic hydrocarbons) like Rhodococcus, Cellulomonas, Nocardioides and Pseudomonas increased (Pyrosequencing data).
Rodrigues et al. (2013)	SWCNT	0, 250, 500 mg kg ⁻¹ soil	Sandy loam (USA)	Microcosms - 3, 7, 14 days incubation	SWCNT reduced the bacterial abundance and modified carbon substrate utilization after 3 days but not after 7 and 14 days. Decrease of fungal abundance and modification of the fungal community structure (T-RFLP analysis) at every date of soil exposure to SWCNT.
Jin et al. (2013)	SWCNT	0, 30, 100, 300, 600 and 1000 mg SWCNT kg ⁻¹ soil	Sandy loam (Korea)	Microcosms - 25 days	Decreases of most enzymatic activities and microbial biomass for the higher concentrations (300, 600 and 1000 mg SWCNT kg ⁻¹ soil).
Jin et al. (2014)	SWCNT	0, 30, 100, 300, 600 and 1000 mg SWCNT kg ⁻¹ soil.	Sandy loam (Korea)	Microcosms - 25 days	High concentrations of SWCNTs significantly altered soil microbial community composition. Gram-positive and Gram-negative bacterial and fungal biomass decreased with higher SWCNT concentrations.

Table 3 Review of the effects of nanoscale zero valent iron nanoparticles on soil microbial communities

Reference	NPs	NPs concentrations	Soil	Experiment	Major results
Cullen et al. (2011)	nZVI dispersed with polyacrylic acid	10 g kg ⁻¹ soil	Silt loam soil (UK)	Microcosms -0, 1, 3, 7 and 14 days	nZVI apparently inhibited microbial ammonia-oxidation and stimulated dehydrogenase activity but had minimal influence on hydrolyase activity. The authors found that nZVI reactivity (redox potential) caused confounding effect on the activity measurements and thus no evidence for negative effects of nZVI on ammonia oxidation and dehydrogenase activities was found.
Fajardo et al. (2012)	nZVI	34 g Fe kg ⁻¹ soil	Sandy clay loam	Microcosms -72 h	nZVI treatment increased Archaea, α -proteobacteria and Gram +C abundances but decreased β and γ -proteobacteria abundances measured by FISH. Abundance of denitrifying bacteria (gene <i>narG</i> and <i>nirS</i>) was also reduced in presence of nZVI but <i>gvrA</i> gene copy number increased (qPCR). The expression of these 3 genes was not modified in the nZVI treatment.
Pawlett et al. (2013)	nZVI stabilized with sodium CMC	270 mg Fe kg ⁻¹ soil	3 soils: sandy, loam and clay amended with 3 concentrations of straw (0, 5 et 10 % organic matter) (UK)	Microcosms -4 months	Microbial biomass was decreased by nZVI only in the soils amended with 5 % straw. Microbial community structure (PLFA profiles) was modified in the 3 soils: sandy (at 0, 5 and 10 % OM), loam (0 and 10 % OM) and clay soil (5 % OM). Negative effects on Gram negative bacteria and arbuscular mycorrhizal fungi. Decrease of Multiple Substrate Induced respiration (MSIR) was only observed in the clay soil. No effect was found on basal metabolic rate.
Tilston et al. (2013)	polyacrylic acid (PAA)-coated nZVI	10 g kg ⁻¹ soil	Silt loam soil (UK)	Microcosms -0, 1, 4, 7 and 28 days	nZVI was associated with altered soil bacterial composition (DGGE profiles) and reduced chloroaromatic biodegradation activities as demonstrated by mineralization assay using 2,4-D as a model chloroaromatic.

community structure using DGGE. After 30 days of soil exposure, TiO₂ and gold nanorods induced the highest changes in the structural diversity of bacterial community. The limited effect of TiSiO₄, CdSe/ZnS quantum dots, and Fe/Co magnetic fluid NPs on DGGE profiles was attributed to their zeta potential values reflecting an unstable state. Hence, once added to the soil, they may have interacted with soil components, becoming unavailable to exert toxic effects.

Metal or metal oxide NPs may be responsible for a biodiversity loss and a modification of soil microbial community composition. Although many research focused on the effect of these NPs on bacterial communities, more work is still required to assess the impact of NPs on fungal and archaeal communities.

Impact of carbon nanoparticles (fullerene and CNTs)

Fullerene NPs had no effect on microbial diversity (Tong et al. 2007) or induced only slight modification of *Eubacteria* and *Kinetoplastida* (Protozoans) community structure on DGGE profiles (20 to 30 % of dissimilarity, Johansen et al. 2008). Pyrosequencing data indicated that MWCNT (10 g kg⁻¹ soil) induced an enrichment of potential degraders of recalcitrant contaminants (PAH) *Rhodococcus*, *Cellulomonas*, *Norcardioides*, and *Pseudomonas*, while some bacterial genera like *Derxia*, *Holophaga*, *Opiritutus*, and *Waddlia* were decreased (Shrestha et al. 2013). Using a comparative metagenomic analysis of bacterial communities, Khodakovskaya et al. (2013) found that the diversity and richness of bacterial communities were not affected by MWCNTs, while a significant modification of the bacterial composition was observed. SWCNTs induced a modification of microbial community composition resulting in a decrease of Gram-positive and -negative bacterial biomass and fungal biomass as well (Jin et al. 2014). Rodrigues et al. (2013) reported also a modification of the fungal community structure after 14 days of soil exposure to SWCNT (250 and 500 mg kg⁻¹). Consistent with activity and abundance measurements, carbon NPs can alter soil microbial community structure but only in presence of high concentrations (>250 mg kg⁻¹).

Impact of nanoscale zero valent iron

The impact of nZVI on microbial diversity was investigated using FISH, DGGE, and PLFA analysis. Fajardo et al. (2012) did not report a broad bactericidal effect of nZVI but observed significant shifts in the structure and phylogenetic composition of the soil microbial community after 72 h of incubation. The FISH assays provided evidence that nZVI exerts a selective pressure on the microbial community, promoting the dominance of some microbial groups (*Archaea*, α -*Proteobacteria*, and low G+C Gram-positive bacteria) or the decrease of other ones (β - and γ -*Proteobacteria* and

subclasses). DGGE profiles also indicated a significant modification of bacterial community composition after 28 days in the presence of 10 g kg⁻¹ of nZVI (Tilston et al. 2013). Pawlett et al. (2013) observed that nZVI caused a modification of PLFA profiles in all soil texture tested, but that these effects were modulated by the organic matter content of the soil. These studies suggest that nZVI could induce a significant modification of soil microbial community structure, affecting bacteria, archaea, and fungi populations on the short term (<4 months).

Concluding remarks

NPs toxicity toward microorganisms has been demonstrated in numerous in vitro studies (e.g., Simon-Deckers et al. 2009; Jiang et al. 2009); yet, the assessment of NPs environmental impact is still in its early stages. This synthesis on the effects of NPs on soil microbial communities supports different conclusions.

It has been demonstrated using different methodologies, different indicators, and natural soils, which NPs could have an impact on microbial activities, abundances, and diversity. However, this synthesis highlights that the toxic effects on microbial community are highly dependent on both the NPs considered and the soil properties. In fact, inorganic NPs (metal, metal oxide NPs) may have a greater toxic potential than organic NPs (fullerenes and CNTs) to soil microorganisms. An exception could be the iron oxide magnetic NPs, which exhibit limited negative effects on microbial communities even when high concentrations were applied. Alarmingly, the use of nZVI could have detrimental effects on biodegradative functions of microorganisms in a context of soil remediation. Further research is needed to determine the real efficiency of nZVI treatments in soil and their potential consequences on soil functioning using activity and functional gene measurements.

The soil properties seem to play a key role for the bioavailability of NPs, especially the clay and organic matter content. We strongly encourage to take more into consideration the physicochemical characteristics of the soil used in the experiments (texture, organic matter content, pH...) and to compare the ecotoxicity of NPs in a range of different soils. The identification of soil parameters controlling the bioavailability of NPs is fundamental for a better environmental risk assessment (Cornelis et al. 2014).

It should be noted that the effects of numerous NPs have not been investigated yet or only in a single study (Al₂O₃, CeO₂, quantum dots, SiO₂, SnO₂...) (Fig. 1), whereas a significant amount of these NPs is susceptible to be released to soils (Keller et al. 2013). Some NPs have been more studied than others (Ag, TiO₂, ZnO) (Fig. 1), and paradoxically, these are not necessarily the most produced and used NPs (Keller

et al. 2013; Sun et al. 2014). The overall number of publications on each class of NP remains still limited to date (≤ 6) (Fig. 1), and thus, it is still difficult to generalize the results. More research is needed, especially through experiments using more environmentally realistic concentrations of NPs based on the predicted concentrations in modelization studies and using more realistic exposure conditions. All the literature analyzed in this review assessed the acute toxicity of NPs corresponding to a sudden disturbance due to a unique application of NPs and the monitoring of the response of microorganisms over time. These experiments assessed the short-term sensitivity of microbial communities, but to date, no data are available neither on the long-term effect and the chronic toxicity of NPs nor on the ability of microorganisms to be resilient to NP disturbance over time.

Although this review emphasizes that microbial ecotoxicology is a valuable approach for risk assessment of NPs on soil ecosystem functioning, the use of microorganisms as indicators of the NPs impact in soil is still in its infancy. At least three reasons might be evoked to explain this fact; first, this is a recent concern steadily increasing since 2011 (Table 1), second this is a complex multidisciplinary subject requiring both biological and physicochemical approaches, and third, it suffers from current technical limitation that hampered our knowledge on NPs behavior in soil. The following section details some crucial subjects that must be investigated to improve our understanding of the environmental impact of these materials using microbial ecotoxicology.

Future research needs

Characterization and fate of NPs in soil

It is unadvisable to conduct ecotoxicological studies without analytical data of the used pollutant. To date, NPs have been extensively characterized in spiking suspensions usually prepared in ultrapure water, in aquatic environments, but not directly in the soil, because of the current technical limitations to detect NPs in complex media (Tourinho et al. 2012; Cornelis et al. 2014). To understand bioavailability and toxicity of NPs, more effort are needed to determine the fate of NPs, i.e., the speciation, the mobility, the homoaggregation and heteroaggregation processes, and all the physicochemical and biological transformations that NPs can undergo in the soil. These issues have been addressed in several reviews (see Lowry et al. 2012; Pan and Xing 2012; Cornelis et al. 2014). The crucial point for the assessment of NP toxicity to soil microorganisms relies on the accurate measurement of the bioavailable fraction of NPs in soil. As discussed by Cornelis et al. (2014), current techniques to estimate the bioavailable fraction of NPs in complex environments are unsatisfactory and need further developments.

Evaluation of a realistic exposure of soil to NPs

Currently, most of the data on the effect of NPs on soil microorganisms were obtained under unrealistic exposure conditions, using NP concentrations 50- to 10,000-fold higher than modeled concentrations in soils (Gottschalk et al. 2009; Sun et al. 2014). Few studies used NPs concentrations $\leq 1 \text{ mg kg}^{-1}$ soil and most of them concerned Ag-NP (Table 1). More research using realistic concentrations is needed especially since it has been demonstrated that these low concentrations can still have detrimental effects for soil microbial communities (Hänsch and Emmerling 2010; Colman et al. 2013; Simonin et al. 2014).

Microcosm experiments enable assessment of the impact of NPs under controlled conditions (temperature, soil humidity...) on short-term monitoring. Mesocosms are particularly effective tools to assess the impact of pollutants on ecosystem functioning under realistic conditions and long-term monitoring. Few studies performed experiments in mesocosms and/or under field conditions to simulate more realistic exposures of soils to NPs. Moreover, the addition of NP in suspension prepared in ultrapure water used in most experiments is not a realistic mode of soil exposure. A likely route by which NPs enter soil is as aged NPs through sewage sludge applications for field fertilization. Further research in these directions will bring useful information to set regulations for the manufacture, use, and disposal of NPs.

Resilience of soil microbial communities to NPs disturbance

This review highlights that soil microorganisms are sensitive to acute contamination of NPs. However, because of the lack of long-term experiments, the resiliency of soil microbial communities to such disturbance and the return to the predisturbance activity levels after sustainable changes in microbial abundance and diversity along time remains still unknown. Allison and Martiny (2008) demonstrated that the composition of most microbial groups is sensitive and not immediately resilient to disturbance, regardless taxonomic breadth of the group or the type of disturbance. This also appears to be true for NP contamination, because in several studies, a greater effect was observed over time, but in period generally not exceeding 2 months (e.g., Ge et al. 2011; Colman et al. 2013). The response of soil microorganisms to NPs needs to be monitored over a long-term periods to evaluate whether ecosystem functioning is permanently disturbed or not (Fig. 2).

From a more realistic point of view, chronic contamination of NPs should be simulated. To date, no study using chronic addition of NPs is available. It is likely that NPs are added to soil chronically through multiple applications of sewage sludge over time. Microbial communities are likely to respond differently to chronic and acute exposures. Acute exposures

may lead to more resistant microbial communities to NP perturbation if sensitive communities are replaced by resistant communities. On the contrary, chronic exposures may have more detrimental effects if microorganisms are not resilient to such repeated contaminations. Studying if resilience of soil microbial community occurs after acute or chronic contaminations with NPs is crucial to evaluate possible consequences on soil functioning over time (Fig. 2).

Effect of mixture toxicity on soil microbial communities

There is a growing interest in the impact of mixture toxicity in soil since most pollutants share some identical routes of entry in soil (wastewater irrigation, sewage sludge application...) and occur simultaneously in polluted sites. With the increasing use of manufactured NPs, it is necessary that NPs are considered in mixture toxicity studies along with heavy metals, hydrocarbons, or antibiotics. Moreover, it is even more crucial because some NPs are able to mobilize some pollutants. In a context of remediation, this ability may be beneficial, but when not controlled, NPs may enhance the bioavailability of other pollutants and caused detrimental effects.

Assessing the impact of NPs in soil exhibiting different physicochemical properties

Soil is a complex matrix and extrapolation of results from one contaminated soil to another is difficult because of the great variability in soil composition and structure (Ranjard et al. 1997). However, most experiments were conducted on a single model soil (Table 1), which does not permit a comparison of soil sensitivity to NPs. Soil properties, such as pH, texture, or organic matter content, influence microbial community composition (Fierer and Jackson 2006) and bioavailability of pollutants (Giller et al. 1998). Thus, the knowledge of the soil physicochemical properties influencing NPs bioavailability (Fig. 2) will greatly enhance our understanding of NPs impact on soil functioning. In addition, we have little information on the spatialization and mobility of NPs in soil (Fang et al. 2009; Vittori Antisari et al. 2013), but like heavy metals, hot spots of pollutants may occur (Ranjard and Richaume 2001). Micro-scale approaches are needed to determine if NPs have a heterogeneous impact in soil. Moreover, the use of microbial indicators may be useful to have better insights on NPs spatialization to circumvent the current lack of techniques to characterize NPs in situ (Tables 2 and 3).

Assessing fungal and archaeal response to NPs

More attention is classically given to soil bacterial communities, despite the crucial role of fungal communities in energy flow and nutrient transfer in terrestrial ecosystems. New insights regarding the broad distribution and abundance of

archaea in soils and oceans imply that they also contribute to global energy cycles, especially nitrogen cycle where ammonia-oxidizing archaea play a key role in nitrification (Schleper et al. 2005; Robertson et al. 2005; Zhang et al. 2010). What is known on NPs toxicity from in vitro studies using bacterial strains may not be extrapolated to archaeal and fungal communities. Thus, it is important to consider the response of archaeal and fungal communities to NPs contamination, in order to have a better assessment of NPs ecotoxicity in soils.

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5. Questions de la thèse

Cette synthèse bibliographique a permis de mettre en évidence les avancées récentes et les verrous qui restent à lever pour mieux comprendre le devenir et l'impact des nanomatériaux dans les sols. Afin d'évaluer les risques liés à deux NPs d'oxydes métalliques (TiO_2 et CuO), nous nous sommes intéressés en parallèle (i) à leur devenir, en considérant leur transport et leurs caractéristiques physico-chimiques dans les sols, et (ii) à leur écotoxicité, en évaluant leur impact sur l'activité, l'abondance et la diversité des communautés microbiennes du sol. Plus particulièrement, les questions auxquelles nous avons souhaité répondre dans cette thèse sont les suivantes :

- Quelle est l'influence des propriétés du sol, notamment la texture et la teneur en matière organique, sur la mobilité des NPs d'oxydes métalliques ?
- Quelle est l'influence des propriétés du sol, en particulier de la texture et de la teneur en matière organique, sur la toxicité des NPs d'oxydes métalliques sur les communautés microbiennes du sol?
- Quel est l'effet de ces NPs sur les groupes fonctionnels microbiens nitrifiants et dénitrifiants impliqués dans le cycle de l'azote, dont les niveaux de redondance fonctionnelle sont différents?
- Les faibles concentrations ont-elles des effets négatifs sur le fonctionnement microbiologique du sol ?
- Existe-t-il une relation dose-réponse associée à l'impact des NPs sur les communautés microbiennes du sol ?
- Le transport et la toxicité des NPs dans les sols sont-ils modifiés lors d'apports chroniques et aigus ?

6. Méthodologie générale

Afin de répondre à ces questions, des choix méthodologiques ont été nécessaires. Ils sont présentés et justifiés ci-dessous.

a. Les nanoparticules d'oxydes métalliques

Nous avons travaillé avec des NPs pures de TiO_2 et de CuO (Figure 13). Comme dit précédemment, le TiO_2 est le nanomatériau le plus représenté dans les sols en raison de son utilisation dans de nombreux produits de grande consommation (cosmétiques, peintures, aliments, médicaments...). Contrairement aux TiO_2 -NPs, les CuO -NPs sont moins fréquemment utilisées et sont donc vraisemblablement présentes dans les sols en plus faibles concentrations. Cependant, elles présentent des propriétés antimicrobiennes bien connues (produits anti-fouling, antibactériens...) et présentent l'un des plus forts potentiels toxiques *in vitro* comparés aux autres NPs (Baek and An 2010 ; Li et al. 2012). C'est pourquoi, il nous a paru intéressant de les prendre en compte dans cette thèse.

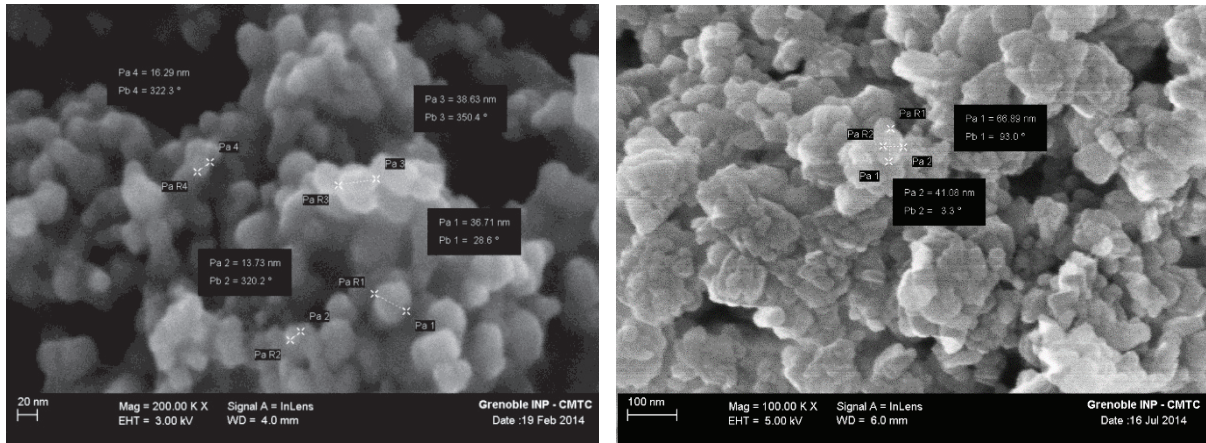


Figure 13: Images de microscopie électronique à balayage (SEM-FEG, Zeiss, CMTC, G-INP) des nanoparticules de TiO_2 (à gauche) et de CuO (à droite) utilisées lors de la thèse.

b. Les sols

Nous avons choisi de travailler avec 6 sols agricoles présentant des propriétés physico-chimiques contrastées en termes de texture (teneur en argile, limon et sable) et/ou de teneur en MO. Trois classes texturales ont été étudiées (sablo-limoneuse, limoneuse et limono-argileuse). Pour chacune d'elles, deux sols ayant des teneurs en MO différentes (i.e.

faible ou forte) ont été prélevés sur un même site. Cette sélection de sols, nous a permis de tester l'influence de 3 textures contrastées sur le transport ou la toxicité des NPs et pour chaque texture, d'apprécier l'influence de la teneur en MO. Les historiques d'utilisation de ces sols sur plusieurs dizaines d'années sont connus car ils font partie de stations expérimentales suivies au laboratoire ou font l'objet d'un suivi dans le cadre du réseau de mesures de la qualité des sols (RMQS – GIS Sol).

c. Les dispositifs expérimentaux

L'étude de la mobilité des NPs de TiO_2 et du CuO a été effectuée en utilisant une approche classique en colonnes de sols saturés en eau dans lesquelles sont injectées en entrée les NPs selon une fonction connue (Figure 14). La fonction de sortie ou courbe de percée est déterminée en analysant les effluents de la colonne, ce qui permet de déterminer la proportion de NPs transportées dans la colonne de sol étudiée en comparaison avec la quantité injectée dans le sol.

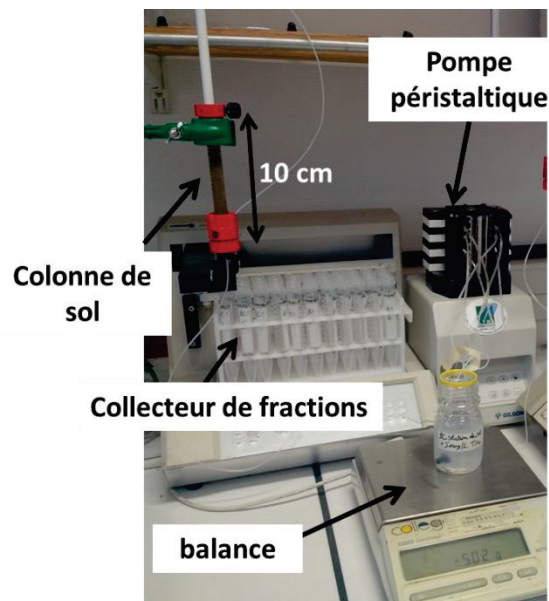


Figure 14: Dispositif expérimental utilisé pour étudier le transport des nanoparticules dans des colonnes de sol (Photo : Marie Simonin). Les colonnes sont en verre et mesurent 1 cm de diamètre et 10 cm de long (Ge-Healthcare Pharmacia Biotech Inc.).

L'étude de la toxicité des NPs de TiO_2 et CuO a été réalisée, dans la plupart des expérimentations, en microcosmes de sols qui ont été contaminés de façon homogène (Figure 15). Cette approche a permis de travailler en conditions contrôlées de température, d'humidité et d'obscurité sur des durées d'exposition allant jusqu'à 90 jours.



Figure 15: Incubation de microcosmes (flacon plasma 150 mL) contenant 50 g sol sec dans une étuve à 28°C (Photo : Marie Simonin).

d. Les indicateurs microbiens

La toxicité des NPs de TiO_2 et du CuO a été déterminée en étudiant l'activité, l'abondance et la diversité des communautés microbiennes qui sont des indicateurs pertinents et complémentaires pour évaluer la qualité d'un sol et sa réponse à une perturbation. Notre but était de déterminer si les NPs pouvaient altérer le fonctionnement biologique du sol (i.e. les activités microbiennes dans notre cas) et, le cas échéant, de comprendre pourquoi le processus mesuré était affecté, en s'intéressant aux effets sur l'abondance, la structure des communautés et la diversité des groupes fonctionnels impliqués (Article 1 : Figure 2).

Nous nous sommes donc intéressés à des processus microbiens globaux et restreints (Schimel *et al.*, 2005 : broad and narrow processes). Nous avons retenu, comme indicateur global, la respiration du sol (ou minéralisation du carbone), qui est réalisée par une grande diversité de microorganismes aérobies et hétérotrophes. Elle est mesurée comme un processus unique par dégagement de CO_2 mais est en réalité la somme d'une multitude de processus impliqués dans la dégradation de la matière organique du sol, qui représente le flux majeur de carbone de ces écosystèmes. Afin de comprendre les effets observés sur la respiration du sol (mesurée par la technique SIR, substrate-induced respiration), nous avons étudié en parallèle les effets sur l'abondance et la diversité bactérienne totale du sol. Les effets sur l'abondance et la diversité des communautés fongiques n'ont pas été pris en compte dans cette thèse.

Des processus microbiens restreints impliqués dans le cycle de l'azote ont été considérés : la nitrification (oxydation de NH_4^+ en NO_3^-) et la dénitrification (réduction séquentielle du NO_3^- en N_2). Ces processus sont réalisés par des groupes de microorganismes qui sont

responsables seulement d'une part du cycle total de l'élément impliqué. Ces groupes fonctionnels sont généralement peu diversifiés et représentent une faible proportion de la communauté microbienne totale (Schimel and Schaeffer, 2012). Par exemple, la nitrification repose sur 2 étapes qui sont la nitritation et la nitratisation (Figure 16). Chacune de ces étapes est réalisée par des groupes fonctionnels microbiens distincts. La nitritation est réalisée par les archées et les bactéries oxydatrices de l'ammonium (AOA et AOB, respectivement), alors que la nitratisation est effectuée dans les sols par des bactéries oxydatrices du nitrite (NOB) appartenant aux genres *Nitrobacter* et *Nitrospira*.

Notre démarche a consisté à évaluer les effets des NPs sur les activités potentielles de nitrification et de dénitrification, ainsi que sur l'abondance et la diversité des groupes fonctionnels impliqués dans ces deux processus en ciblant des gènes de fonction.

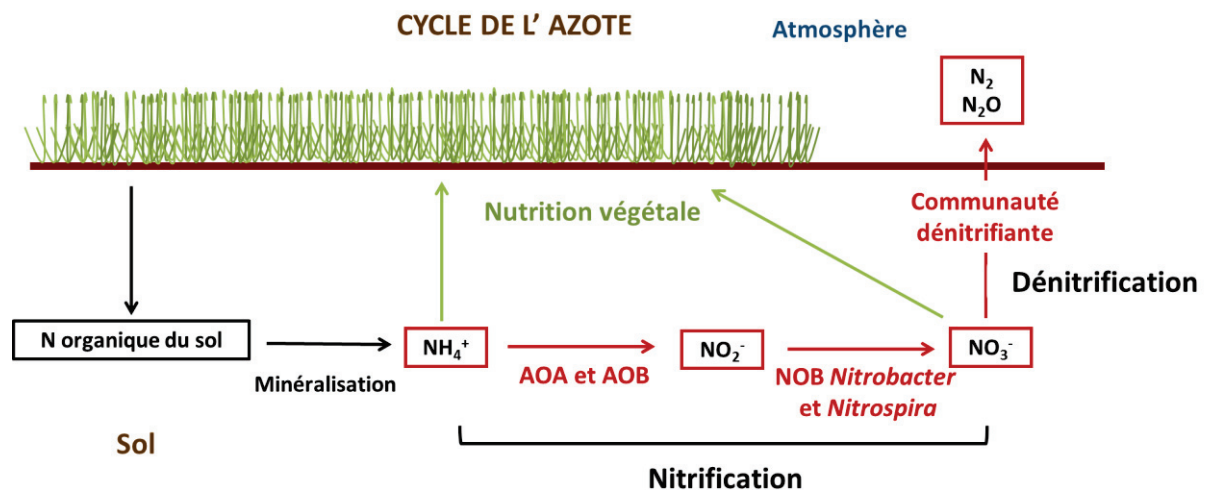


Figure 16: Schéma simplifié du cycle de l'azote dans les sols. Les groupes fonctionnels microbiens impliqués dans la nitrification et la dénitrification sont indiqués en rouge.

L'étude des activités microbiennes (respiration, nitrification et dénitrification) nous a permis d'évaluer l'impact des NPs sur des processus clés des cycles du carbone et de l'azote, essentiels pour la fertilité des sols. Cela nous a également permis de déterminer si l'impact des NPs était différent sur des activités microbiennes reposant sur des groupes fonctionnels possédant des degrés de redondance fonctionnelle différents (le nombre d'espèces capables de réaliser la même fonction). En se basant sur l'« insurance hypothesis » proposée par Loreau *et al.*, (2002), on s'attend à ce qu'une plus grande diversité au sein d'un groupe fonctionnel confère une meilleure résistance du processus associé à une perturbation environnementale. Cette hypothèse se base sur l'idée intuitive que la probabilité de trouver

Chapitre 1: Synthèse bibliographique, questions et choix méthodologiques

des espèces capables de s'adapter lors de perturbations et contribuant à la fonction est plus grande dans un groupe ayant une forte diversité. Dans notre cas, on pourrait donc s'attendre à ce que le processus le plus sensible aux NPs soit la nitrification (faible redondance) et le plus résistant, la respiration du sol (forte redondance).

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

1. Introduction

Il existe une grande diversité de sols en termes de texture, structure, pH ou de teneur en MO par exemple. Ces propriétés physico-chimiques variées vont influencer le devenir des NPs dans les sols. Ces propriétés influencent également la composition et la physiologie des communautés microbiennes du sol. De ce fait, on peut faire l'hypothèse que le devenir et l'impact des NPs va différer d'un sol à un autre en fonction de sa composition physico-chimique et de la communauté microbienne qu'il héberge.

Toutefois, à l'heure actuelle peu d'études ont étudié le devenir des NPs dans différents types de sols naturels et la majorité des études d'écotoxicologie microbienne a été réalisée sur un seul sol modèle (Article 1). Dans le cadre de l'évaluation des risques associés à ces polluants émergents dans les écosystèmes terrestres, il serait nécessaire d'identifier des paramètres physico-chimiques clés du sol qui influencent les propriétés des NPs et donc leur mobilité et leur toxicité.

Nous nous proposons donc dans ce chapitre d'évaluer l'influence de la texture et de la teneur en MO du sol sur la mobilité (Article 2) et la toxicité des NPs (Article 3). Pour cela, des approches expérimentales en colonnes et en microcosmes ont été mises en place en étudiant 6 sols agricoles aux propriétés physico-chimiques contrastées.

2. Influence des propriétés du sol sur le transport du TiO₂ et CuO en colonnes de sol

a. Article 2 : Présentation générale de l'étude et synthèse des principaux résultats

La majorité des études disponibles sur le transport des NPs a été réalisée dans des milieux poreux modèles (sable, billes). Ces études ont montré l'importance de considérer l'influence de facteurs abiotiques, tels que la MO, la force ionique, l'argile, pour comprendre la mobilité de ces polluants. Toutefois, ces milieux poreux modèles ne représentent pas les conditions complexes retrouvées dans des sols naturels, en termes de variété des types de surfaces minérales et organiques, de chimie de la solution du sol, ou de granulométrie.

C'est pourquoi, dans cette étude nous avons étudié le transport des TiO₂ et CuO-NPs dans des colonnes de 6 sols naturels présentant des textures et teneurs en MO différentes. Les concentrations en titane et cuivre retrouvées dans les effluents des colonnes ont été mesurées après digestion acide et quantification par ICP-OES, afin de déterminer la part mobile des NPs et la part retenue dans le sol.

Nous avons mis en évidence que les CuO et TiO₂-NPs avaient une mobilité faible à très faible dans les 6 sols étudiés, avec 86 à 98% des NPs retenues dans les colonnes de sol. Ni la texture ni la teneur en matière organique n'ont d'effet notable sur le transport du TiO₂. En revanche, pour les CuO-NPs dans les sols sablo-limoneux et limono-argileux le transport était contrasté entre les sols avec faible ou forte teneur en MO. Dans le sol sablo-limoneux, la mobilité du CuO était plus importante dans le sol présentant une forte teneur en MO, alors que l'inverse était observé dans le sol limono-argileux. Ces différences de mobilité ne semblent pas liées à des différences d'agrégation ou de charge de surface du CuO-NPs mais plutôt à une forte concentration en carbone organique dissout dans la solution du sol qui pourrait favoriser le transport des NPs.

Nos résultats indiquent que la mobilité des NPs d'oxydes métalliques, évaluée sur une distance de 10 cm, est très limitée et donc que le transfert de ces polluants vers les couches profondes du sol ou les eaux souterraines serait négligeable. Nous n'avons pas pu mettre en évidence d'effet en lien avec la porosité des différents sols car dans les sols sablo-limoneux et limono-argileux, les taux de rétention des NPs étaient similaires.

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

b. «Influence of clay and organic matter content on the transport of TiO₂ and CuO nanoparticles in saturated soil columns»

L'article 2 intitulé « Influence of clay and organic matter content on the transport of TiO₂ and CuO nanoparticles in saturated soil columns » sera soumis dans le journal *Environmental Pollution*.

Influence of clay and organic matter content on the transport of TiO₂ and CuO nanoparticles in saturated soil columns

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ABSTRACT

The transport of metal-oxide nanoparticles (CuO and TiO₂) in 6 natural soils was studied in saturated columns. In particular, we investigated the influence of clay and organic matter (OM) content on their transport. For this purpose, breakthrough curves of CuO and TiO₂ were measured in soils belonging to 3 different textural classes (sandy-loam, loam and silty-clay) and for each of them, two contrasted OM content.

CuO was mobile in all soils but the retention was generally high (86% minimum). No clear influence of soil texture could be observed on CuO transport, but in the sandy-loam and silty-clay soils, it was affected by the OM content. In all soil solutions, CuO dissolution was negligible (0.14 to 0.49 %) suggesting that the Cu concentration measured in the effluent could mainly be attributed to CuO nanoparticles and not to its dissolved form. The TiO₂-NPs mobility in all soil columns was found to be very low with a mass recovery value of 1.5 – 5.5% and was always lower than CuO, which may be related to the higher homoaggregation of TiO₂ in soil solutions. Due to its high retention in soil, no clear influence of soil texture or OM content could be observed on the TiO₂ transport.

Our result show that CuO and TiO₂ nanoparticles exhibit a low mobility in a range of natural soils and that dissolved organic carbon is certainly a key driver of CuO mobility in soil.

Key-words: Soils, Nanomaterials, Transfer, Dissolved organic carbon, CDE modelling

INTRODUCTION

Metal oxide engineered nanoparticles (NPs) are becoming integral elements of a myriad of consumer and industrial products. Copper oxide (CuO) NPs are used extensively in catalysts, superconductors or in paints for their biocidal properties (Zhu *et al.*, 2004; Ren *et al.*, 2009; Keller *et al.*, 2013). Titanium dioxide (TiO₂) NPs are commonly integrated in personal care products, paints or food packaging (Keller *et al.*, 2014). As a result, soils are exposed to these components through various exposure routes, which may lead to potential risks for humans and ecosystems health. Their release into soils can occur during industrial production, landfills, agricultural amendments of sewage sludge, abrasion of materials and accidental spill (Nowack *et al.*, 2012). Therefore, it appears urgent to improve our current understanding of the fate and transport of NPs in soils in order to better assess the risks for groundwater and agricultural soils.

Transport of NPs has been largely investigated in well-defined simple porous media, such as glass beads or quartz sand, but few investigations have addressed the transport of metallic NPs in natural soils (Fang *et al.*, 2009, 2011; Cornelis *et al.*, 2013). Soils are porous systems consisting in complex structured assemblies of mineral and organic particles combined with liquid and gaseous phases. Several soils components such as clay and organic matter (OM) are known to influence the mobility of colloids because of their important reactive surfaces, which can interact with particles and modify their physicochemical behavior (Pan and Xing, 2012). Several studies have demonstrated the influence of these factors on NPs aggregation, surface charge and stability (Fang *et al.*, 2009; Thio *et al.*, 2011; Zhou *et al.*, 2012). Dissolved OM is known to favor NPs dispersion and thus NPs mobility in simple porous media (Ben-Moshe *et al.*, 2010; Thio *et al.*, 2011), while clay particles can destabilize positively and negatively charged NPs and thus decrease their mobility (Zhou *et al.*, 2012). Moreover, these processes are greatly influenced by the chemistry of the soil solution, such as pH, ionic strength and electrolytes (Wang *et al.*, 2012; Cornelis *et al.*, 2014; Vitorge *et al.*, 2014). Fang *et al.*, (2011) demonstrated that TiO₂-NPs could act as a metal carrier and facilitate for instance the mobilization of Cu in soil. These results raise some concern about the potential environmental risk associated with enhanced migration rates of co-contaminants and the increased bioavailability of some toxic metals in presence of NPs. Due to the interactions between these multiple environmental factors in natural soils, it is difficult to predict the

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

fate of NPs in soil from investigations conducted in simple porous media. In particular it remains highly difficult to hierarchize the dominant factors controlling the transfer of manufactured NPs in natural soils, since most of these factors are closely linked. Studies evaluating the simultaneous effects of several soil factors on NPs mobility are clearly needed to elucidate the key soil properties controlling NPs mobility.

The objectives of this study were to determine the influence of soil texture and OM content on the transport in natural soils of two relevant manufactured NPs, CuO and TiO₂. We conducted saturated column experiments with 6 agricultural soils of contrasted physicochemical properties: 3 textures: sandy-loam, loam and silty-clay, and 2 different OM contents for each soil texture. The NPs breakthrough curves were fitted with the Convection Dispersion Equation (CDE) in order to identify the dominant processes involved in these NPs transfer in soil and quantify the corresponding parameters.

MATERIALS AND METHODS

Soils

This study was conducted with soils belonging to three different textural classes: a sandy-loam, a loam and a silty-clay. Soils were collected from the upper 20 cm layer of 3 different agricultural sites located in the Rhône-Alpes and Burgundy regions of France (see Article 3: Simonin *et al.*, 2015 for further details). At each site, the sampling was performed in 2 plots with low and high OM contents distant of less than 500 m. Therefore, the experiments were performed with 6 test soils: with 3 different textures and 2 different OM contents for each texture. After collection, visible rocks, roots and plant litter were manually removed. The soils were sieved (2 mm) and homogenized before storage at 4°C. Soils were previously characterized (Simonin *et al.*, 2015). Some selected soil physicochemical properties are presented in Table 1.

Soil solutions of the 6 soils were prepared following Simonin *et al.*, (2015) and used as background solutions during the column experiments. Briefly, soil solutions were prepared by shaking 10 g of soil dispersed in 50 mL of ultrapure water (18M Ω) during 30 minutes at 180 rpm and 20°C in a refrigerated incubator shaker (New Brunswick - Eppendorf, Hamburg,

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

Germany). The solutions were then centrifuged for 20 min at 8000 g, 20°C (Centrifuge 5804R, Eppendorf, Hamburg, Germany) to eliminate particles larger than 20 nm according to the Stokes' law. The supernatants were collected and stored at 4°C before use in column experiments.

Table 1 Main characteristics of the six studied soils

	Soils										Soil solutions		
	% sand	% loam	% clay	% OM	% WHC	pH	Ionic strength (mM)	CEC (cmol ⁺ kg ⁻¹)	Cu (mg kg ⁻¹)	Ti (g kg ⁻¹)	pH	Ionic strength (mM)	DOC (mg L ⁻¹)
Sandy-loam - Low OM	68.4	14.7	16.9	2.09	20	7.0	0.98	11.5	20.1	2.4	6.7	1.0	11.6
Sandy-loam - High OM	65.6	16.1	18.3	4.46	20	6.9	1.59	15.4	21.9	2.4	6.4	2.0	18.6
Loam - Low OM	37.5	42.7	19.8	2.23	30	6.4	0.60	8.79	13.2	2.7	6.2	1.2	8.2
Loam - High OM	40.3	40.8	18.9	6.77	30	6.3	1.31	15.3	10.4	2.4	5.4	2.4	10.5
Silty-clay - Low OM	8.2	49.8	42.0	4.72	47	6.9	0.51	17.4	23.2	5.3	6.3	1.5	25.2
Silty-clay - High OM	10.1	50.8	39.1	7.87	51	7.7	1.37	20.1	26.2	4.5	7.1	1.6	8.9

Nanoparticles

TiO₂-NPs were provided by Sigma Aldrich (St Louis, USA) as a mixture of anatase (80%) and rutile (20%) crystal structure with at least 99.5% purity. According to the manufacturer information, TiO₂-NPs presented a specific surface area of 35–65 m² g⁻¹ and a mean particle size of 21 nm in powder as measured by Transmission Electron Microscopy. CuO-NPs were purchased from Sigma Aldrich (St Louis, USA) with a nominal size < 50 nm and a specific surface area of 23 m² g⁻¹, according to the manufacturer information. Both NPs were characterized for intrinsic primary particle size using a ZEISS Ultra 55 scanning electron microscopy – field emission gun (SEM-FEG). In average, TiO₂-NPs measured 28.7 ± 7.1 nm and CuO-NPs measured 57.0 ± 18 nm.

The apparent hydrodynamic diameter and zeta potential of the NPs were characterized using Dynamic Light Scattering (DLS) in the soil solutions used as background solutions. The measurements were performed in triplicate in the spiking suspensions of NPs (soil solution + NPs at 50 mg L⁻¹), after dispersion through ultrasonication for 5 min to ensure suspensions homogeneity.

Column experiments

The transport experiments were performed in small glass columns (C10/10, GE Healthcare) of 1 cm in diameter and 10 cm long, homogeneously packed with 8 cm of wet soil. Flow adaptors (AC 10, GE Healthcare) were adjusted on the top of the columns to ensure a constant soil height during experiments. At the beginning of the experiments, soil columns were saturated and leached with 100 mL of soil solution. The effluents were collected using a fraction collector to check for the turbidity of the outflow by measuring the absorbance at 600 nm (Spectrometer Biowave II, Biochrom WPA). The absorbance was very low for all soils (< 0.02), suggesting that the amount of soil colloids in the effluents was negligible. Bromide (Br) ion was used as water tracer to assess the hydrodynamic properties of the soil columns before NPs injection. One pore volume of Br tracer (KBr 1 g L^{-1} prepared in soil solution) was injected in each soil column and followed by 2 pore volumes injection of soil solution. The concentrations of Br in the soil column effluents were measured by Ion chromatography (Metrohm 732/733 separation center) with a detection limit (DL) of $4 \mu\text{g L}^{-1}$.

The spiking suspensions of NPs (TiO_2 or CuO) were prepared at 50 mg L^{-1} (C_0) in soil solution before the beginning of the experiment and were injected in the columns at a flow rate of 0.1 mL min^{-1} during 5 pore volumes. After NPs spiking, the columns were continuously leached with the soil solution for at least 15 pore volumes. The column effluents were sampled in 15 mL centrifuge tubes every 10 minutes ($\sim 1 \text{ mL}$). Titanium or copper concentrations were measured in the spiking suspensions (C_0) and in the effluents (C) to construct breakthrough curves of NPs presented in a dimensionless form: C/C_0 as a function of V/V_0 , the dimensionless eluted volume, V , normalized by the total water volume of the column, V_0 . The NPs analysis were performed using microwave assisted (Novawave, SCP Science) strong acid extraction (for titanium: hydrofluoric acid + nitric acid and for copper: *aqua regia*). Elements concentrations were determined using inductively coupled plasma optical emission spectrophotometer (ICP-OES; Varian 700-ES, Varian Inc. Scientific Instruments, Palo Alto, USA). With this methodology, titanium and copper concentrations measured in a certified soil reference material were in good agreement with certified values (data not shown). The detection limit was $50 \mu\text{g L}^{-1}$ for both Cu and Ti measurements. Control columns without NPs injection were performed for each soil to determine the

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

background concentration of titanium and copper in the effluents. The physicochemical properties of the soils columns are provided in Table 2.

To evaluate the occurrence of CuO dissolution in spiking suspensions during each column experiment, a dissolution test was performed. The 6 spiking suspensions (one for each soil) were prepared and sampled (5 mL) after 6 hours corresponding to the maximum duration of the NPs column transport experiments. Dissolved Cu concentration in samples were determined by ICP-OES (Varian 700-ES) after centrifugation for 20 min at 6000 g in a 5 kDa ultrafiltration device (Vivaspin tube, Sartorius) in triplicate. Soil solutions without CuO-NPs were used as controls. The dissolution of TiO₂-NPs in spiking suspensions was not assessed because these NPs are not soluble in water (Chen and Mao, 2007; Duester *et al.*, 2011).

Table 2 Main characteristics of the soil columns

	Pore volume (mL)	Bulk density (g cm ⁻³)	Effective Porosity	Darcy velocity (cm h ⁻¹)	Average soil collector diameter (μm)
Sandy-loam - Low OM	4.3	1.27	0.51	0.38	122.9
Sandy-loam - High OM	4.8	1.08	0.59	0.38	118.3
Loam - Low OM	5.1	0.83	0.68	0.25	74.5
Loam - High OM	5.4	0.86	0.67	0.25	78.9
Silty-clay - Low OM	5.9	1.01	0.61	0.16	24.9
Silty-clay - High OM	6.7	0.94	0.64	0.15	28.4

Transport parameters estimation

The CXTFIT code implemented in the STANMOD software (USDA) (Toride *et al.*, 1995) was used to model tracer and NPs transport experiments. The pore water velocity (V , cm h⁻¹) and dispersion (D , cm² s⁻¹) in the NPs transport simulations were determined from the Br tracer breakthrough curves. The analytic solution of the Convection Dispersion Equation (CDE) with a first-order loss term presented in Eq.1 was fitted to the Br and NPs breakthrough curves using the Levenberg Maquardt algorithm (e.g. Martins and Mermoud, 1998; Martins *et al.*, 2013; Vitorge *et al.*, 2013; Lakshmanan *et al.*, 2015).

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial z^2} - V \frac{\partial C}{\partial z} - \mu C \quad \text{Eq.1}$$

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

where C is the concentration, t is time, z is distance from the inlet, V is average pore water velocity, D is dispersion coefficient and μ is the decay coefficient, related to NPs irreversible retention on solid collectors. For Br experiments V , R (retardation factor, dimensionless) and D were calculated. For NPs experiments the parameters V , R , D and μ were calculated for the fitting of the breakthrough curves.

Statistical analysis

The results of NPs hydrodynamic diameter, zeta potential and dissolution in soil solutions are presented as means (\pm standard errors) of 3 replicates. The influence of soil texture and OM content on NPs hydrodynamic diameter and zeta potential was investigated with a two-way Anova using the R software.

RESULTS AND DISCUSSION

Physicochemical properties of nanoparticles in soil solutions

CuO and TiO₂-NPs were characterized in the 6 soil solutions used as spiking suspensions during the transport experiments (Figure 1). The TiO₂-NPs hydrodynamic diameter values ranged from 73.9 to 128.1 nm and a significant effect of OM content and of the interaction of OM content and soil texture was observed on this parameter (Figure 1A, Table 3). These significant effects were related to the higher aggregation of TiO₂-NPs in the silty-clay soil (High OM). This higher aggregation could be attributed to the low DOC content and the high concentration of Ca in relation with the alkaline pH of this soil solution (Table 1, Figure 1A) (Simonin *et al.*, 2015; Badawy *et al.*, 2010; French *et al.*, 2009). The CuO-NPs hydrodynamic diameter values ranged from 53.3 to 84.1 nm, meaning that almost no aggregation of CuO-NPs occurred in soil solution because their diameter in powder was 57 ± 18 nm in average. Thus, CuO-NPs size was lower than TiO₂-NPs in all soil solutions, except in the loam soil (High OM) (Figure 1B). No significant influence of soil texture or OM content on CuO-NPs aggregation was observed (Table 3).

Both TiO₂ and CuO-NPs were negatively charged in all soil solutions (Figure 1C and 1D) and these zeta potential values were significantly affected by soil texture and OM content (Table

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

3). TiO₂-NPs zeta potential ranged from -15.5 to -23.9 mV and these zeta potential values in soil solutions with low OM content were always more negatively charged than in high OM content soil. CuO-NPs zeta potential ranged from -13.8 nm to -20.8 mV and conversely to TiO₂-NPs, the CuO-NPs in soil solution with low OM content were always less negatively charged than in high OM content soil, probably in relation with the high affinity of copper for OM (Jacobson *et al.*, 2007; Morel *et al.*, 2014; Navel and Martins, 2014).

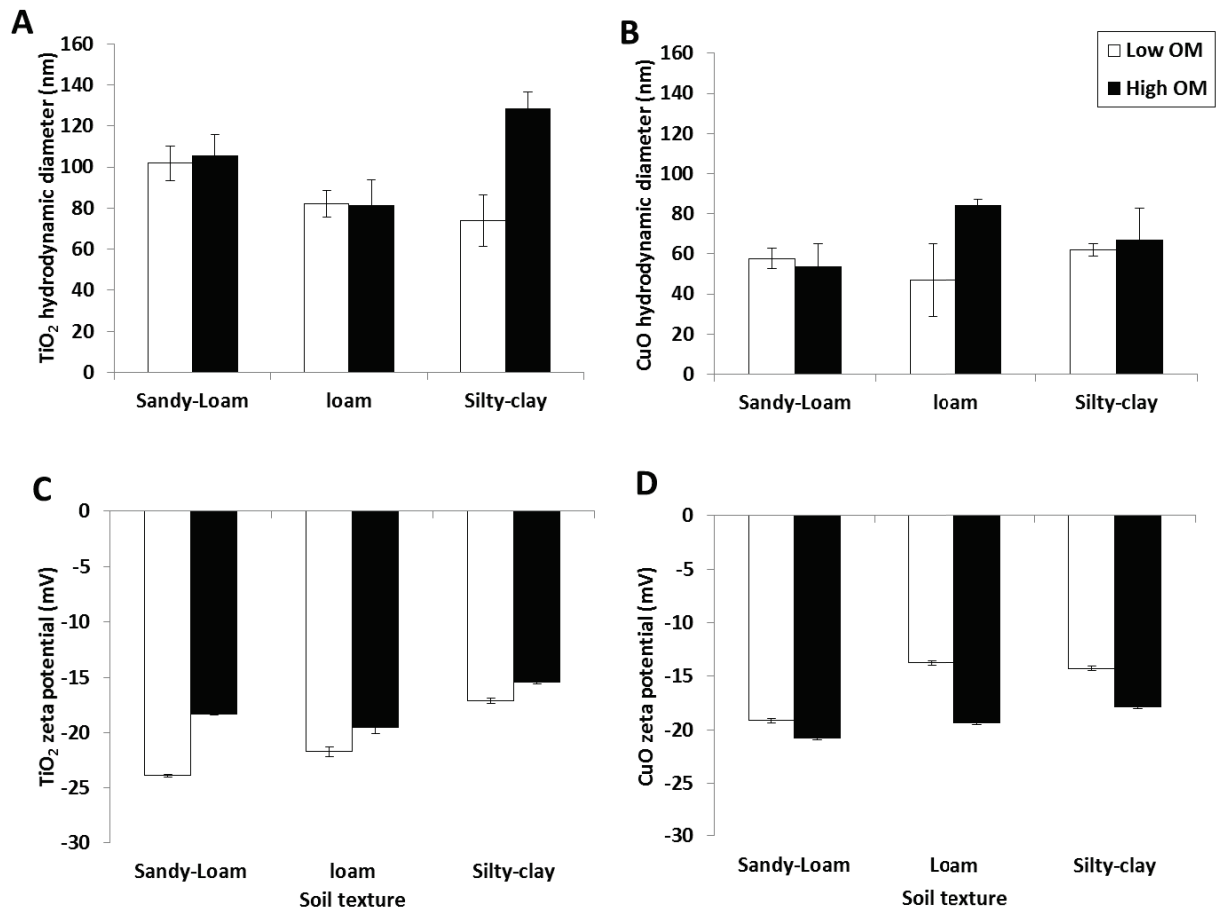


Figure 1 TiO₂ and CuO-NPs hydrodynamic diameter (A and B, respectively) and zeta potential (C and D, respectively) measured in background soil solutions prepared at 50 mg NPs L⁻¹ before transport experiments.

Table 3 *P*-values associated to the effect of soil texture, OM content and the interaction of texture and OM on the size (i.e. hydrodynamic diameter) and zeta potential of TiO₂ and CuO-NPs.

	TiO ₂		CuO	
	Size	Zeta potential	Size	Zeta potential
Texture	0.12	< 0.001	0.64	< 0.001
OM content	0.04	< 0.001	0.19	< 0.001
Texture x OM	0.04	< 0.001	0.20	< 0.001

Hydrodynamic properties of the 6 soils

The hydrodynamic properties of the 6 soils were determined by fitting the Br breakthrough curves with the CXTFIT code (Figure 2). The corresponding transport parameters are presented in Table 4 along with those fitted for the CuO-NPs.

All tracer breakthrough curves presented mass recoveries close to 100%, as expected with a water tracer. Retardation (R) factors were all close to 1 except that calculated in the Silty-clay low OM, in which R was 0.82, indicating that part of the water in soil was not seen by the tracer, in relation with the high clay content of this soil. The dispersion values (D) fitted to the breakthrough curves are quite low (below 10^{-7} cm² s⁻¹) and very similar for all soils. These parameter values were used to fit the breakthrough curves of the NPs.

Transport of CuO-NPs in soil columns

The dissolution of CuO-NPs was assessed in the spiking suspensions. CuO dissolution was negligible in all soil solutions, as it ranged between 0.14 and 0.49 % of the NPs applied mass after 6 hours, representing the duration of NPs injection in soil columns. These results show that Cu concentrations measured in the effluents can be attributed to CuO-NPs.

The observed and calculated breakthrough curves of CuO-NPs are presented in Figure 2. The corresponding parameters are presented in Table 4. The breakthrough curves of CuO-NPs show that these NPs were strongly retained in the 6 studied soils with quite low mass recoveries ranging between 5.2 and 14% (Table 4).

No clear influence of soil texture on CuO-NPs transport could be observed. In particular, CuO-NPs transport was not specifically higher in the sandy-loam soils than in the loam and silty-clay soils that have higher clay contents, as it could be expected from previous works (Fang *et al.*, 2009). However, in the sandy-loam and silty-clay soils, the OM content affected

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

the transport of CuO-NPs (Figure 2), while in the loam soils the breakthroughs were very similar in both low and high OM soils (Figure 2). Interestingly in the sandy-loam soil, the mobility increased of 62% in the high OM soil compared to the low OM soil, whereas in the silty-clay soil, CuO-NPs mobility decreased of 44% in the high OM soil compared to the low OM soil (Table 4, Figure 2).

In the sandy-loam (high OM) and the silty-clay (low OM) soils, where CuO-NPs transport was enhanced, the DOC concentration in soil solution was higher than in the other soils (Table 1). Several soil factors may be involved in the lower retention of CuO-NPs in these two soils, however DOC concentration appears as a main driver of CuO-NPs mobility, as already demonstrated in model porous media (Ben-Moshe *et al.*, 2010). The difference of zeta potential of CuO-NPs measured in soil solution do not seem to affect the NPs mobility in soil columns. Indeed in the loam soils, CuO-NPs breakthrough curves were very similar (Figure 2) although their zeta potential values were the most contrasted in the low and high OM soils (Figure 1).

All CuO-NPs breakthrough curves were well fitted with the CXTFIT code, using the hydrodynamic parameters determined with the water tracer. In all soils, the eluted CuO-NPs presented a fast transport in the columns without retardation as compared to the Br tracer (Table 4). In some soils, CuO-NPs presented a fast transport (faster than the tracer) indicating that these particles did not see all the water present in the soil, indicating some size exclusion processes during their transport, in agreement with the colloid filtration theory.

To the authors' knowledge, this is the first study on the transport of CuO-NPs in natural soils. We have shown that CuO-NPs could be partly mobile in very contrasted soils although the majority (86 to 94.2%) of these NPs are retained in the soils. Further work is needed to evaluate precisely the role of DOC in CuO-NPs transport, especially the influence of the type and concentration of natural OM present in the soil solutions.

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

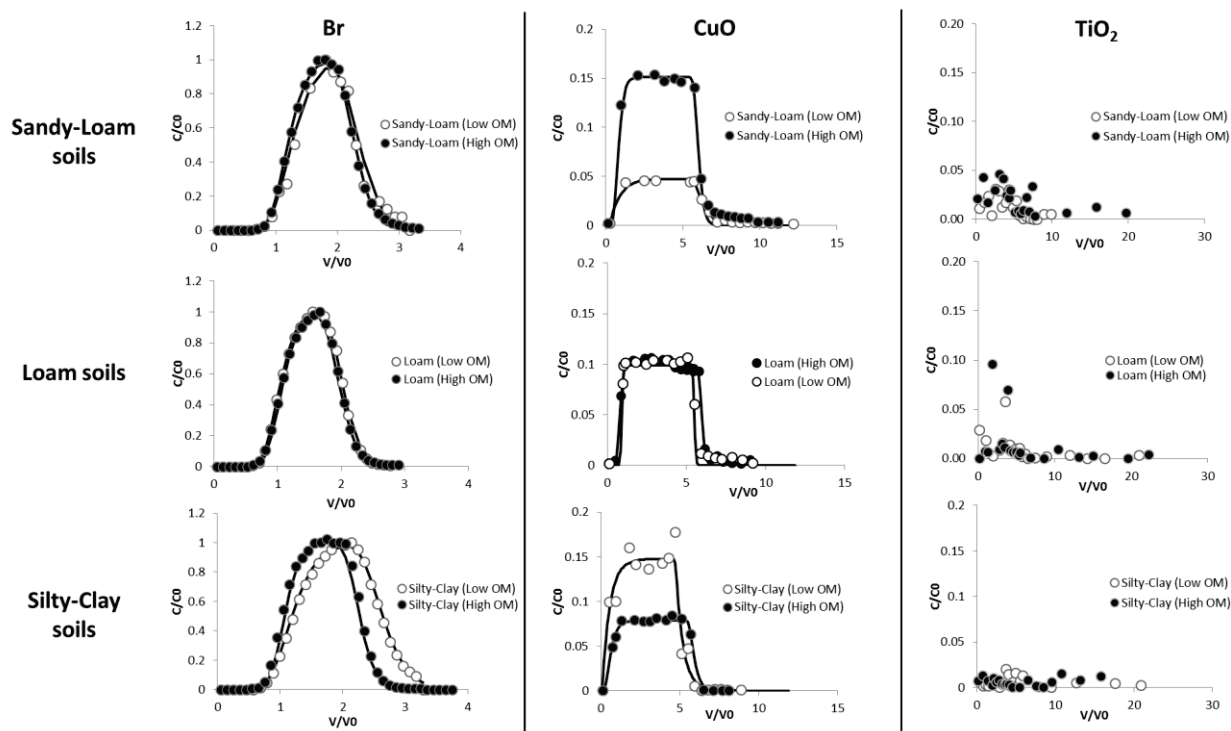


Figure 2 Breakthrough curves of Br, CuO and TiO₂ with CXTFIT predicted Br and CuO concentrations in columns of the 6 soils. Symbols represent the measured data, and black lines were calculated with the CXTFIT code by fitting with STANMOD.

Table 4 Transport parameters for bromide, CuO-NPs and TiO₂-NPs experiments.

		Bromide			CuO-NPs				TiO ₂ -NPs
		MR* (%)	D (cm ² /s)	R (-)	MR* (%)	D (cm ² /s)	R (-)	μ (h ⁻¹)	MR* (%)
Sandy-loam	Low OM	96.2	1.0E-07	1.01	5.2	1.35E-05	1.10	7.27	2.1
	High OM	99.1	7.1E-08	0.94	14.0	2.52E-07	1.01	1.97	5.5
Loam	Low OM	98.4	1.0E-07	0.96	11.4	1.10E-08	0.93	2.29	2.3
	High OM	99.0	5.9E-08	0.96	9.7	8.30E-08	0.87	2.37	3.4
Silty-clay	Low OM	101.3	1.2E-07	0.96	12.1	1.04E-05	1.56	2.37	1.9
	High OM	98.7	5.5E-08	0.82	6.8	9.47E-07	1.23	3.28	2.2

* mass recovery (%)

Transport of TiO₂-NPs in soil columns

The mobility of TiO₂-NPs in all soil columns was found to be very low with mass recoveries ranging between 1.5 and 5.5% (Table 4). In the 6 soils, the breakthrough curves were noisy indicating that the measured Ti concentrations were close to Ti background concentration in the leachates (Figure 2). Consequently, no transport modelling was conducted on TiO₂-NPs breakthrough curves. In all soils, TiO₂-NPs retention was higher than CuO-NPs. This very strong retention probably relates to the higher homoaggregation of TiO₂-NPs in soil solutions (Figure 1) (Vitorge *et al.*, 2013). Contrary to CuO-NPs, no clear influence of OM content on TiO₂-NPs mobility was noticed in the 6 soils. The high retention of TiO₂-NPs in soil is consistent with previous studies (Nickel *et al.*, 2015; Wang *et al.*, 2014) although these experiments were performed with positively charged TiO₂-NPs. As in the present work, Fang *et al.*, (2009) used negatively charged TiO₂-NPs and observed a significant transport of these NPs, especially in soils with low clay content and ionic strength. Since the retention of TiO₂-NPs was very high in the 6 soils of this study, it was not possible to observe a clear influence of the soil clay content (soil texture) or of the ionic strength. The differences with the results of Fang *et al.*, (2009) could be due to the different methodology used to assess TiO₂-NPs transport, especially the NPs input concentration, the column size and setup and the titanium measurement technique in the effluents. Studying the fate of TiO₂-NPs in soil is very challenging as Ti is very abundant in soils and has a heterogeneous distribution. The findings obtained with saturated soil columns carried out in laboratory conditions give a first assessment of the risk of NPs migration to deeper soil layers or groundwater. However, further work is needed to determine the transport potential of NPs with environmentally relevant concentrations, in undisturbed soils or under unsaturated conditions, in order to be closer to expected field conditions (Duester *et al.*, 2011; Cornelis *et al.*, 2014).

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3. Influence des propriétés du sol sur l'impact du TiO₂ sur la respiration microbienne et l'abondance bactérienne

a. Article 3 : Présentation générale de l'étude et synthèse des principaux résultats

Compte tenu de la mobilité différente des NPs modèles en fonction du sol et de la forte rétention observée, nous avons souhaité évaluer (i) si les TiO₂-NPs présentaient une toxicité vis à vis des communautés microbiennes indigènes de ces sols et le cas échéant (ii) déterminer quelles propriétés des sols influençaient cette toxicité.

Les 6 sols ont donc été exposés à deux concentrations en TiO₂ (1 mg et 500 mg kg⁻¹) et incubés en microcosmes pendant 7, 30 et 90 jours. L'impact sur les communautés microbiennes du sol a été évalué en ciblant un processus microbien global du cycle du carbone : la respiration du sol, également appelée minéralisation du carbone, mesurée par la méthode dite de substrate-induced respiration (SIR). En parallèle, les effets sur l'abondance bactérienne ont été mesurés par PCR quantitative en ciblant le gène *16S ADNr* (également appelé *rrs*). Afin de déterminer l'influence des propriétés du sol sur les caractéristiques physico-chimiques des TiO₂-NPs, celles-ci ont été caractérisées dans les solutions de sol et dans de l'eau ultrapure en mesurant leur agrégation (diamètre hydrodynamique) et leur charge de surface (potentiel zêta) par Dynamic Light Scattering à l'aide d'un NanoZS (Malvern).

Aucun impact du TiO₂ n'a été constaté sur la minéralisation du carbone ou l'abondance bactérienne dans les sols, à l'exception du sol limono-argileux à forte teneur en MO. Dans ce sol, une diminution de la minéralisation du carbone a été observée dès 7 jours, même pour la dose faible de 1 mg kg⁻¹ (jusqu'à -20%). Nos résultats suggèrent que la toxicité du TiO₂-NPs pourrait être liée au pH et à la teneur en MO mais pas à la teneur en argile des sols. De plus, nous avons constaté que dans la solution du sol limono-argileux (forte MO), les TiO₂-NPs seraient moins stables en suspension (potentiel zêta entre -15 mV et +15 mV), due à une diminution de la répulsion électrostatique. Cette caractéristique pourrait expliquer leur plus grande toxicité envers les microorganismes de ce sol, mais cela reste à vérifier.

Nos résultats indiquent que, dans la majorité des sols, le TiO₂ a un faible potentiel toxique envers les microorganismes. Dans l'unique sol où des effets délétères ont été observés, les

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

TiO₂-NPs avaient des propriétés physico-chimiques singulières par rapport aux autres sols, indiquant une plus faible stabilité en suspension.

b. « Influence of soil properties on the toxicity of TiO₂ nanoparticles on carbon mineralization and bacterial abundance »

L'article 3 intitulé «Influence of soil properties on the toxicity of TiO₂ nanoparticles on carbon mineralization and bacterial abundance » a été publié en 2015 dans *Journal of Hazardous Material*.



Influence of soil properties on the toxicity of TiO₂ nanoparticles on carbon mineralization and bacterial abundance



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HIGHLIGHTS

- We tested TiO₂-NPs effects on microbial communities in six contrasted soils.
- In all soils but one, TiO₂-NPs had no impact on microbial communities.
- A low dose of TiO₂-NPs decreases C-mineralization in a silty-clay soil.
- TiO₂-NPs toxicity is likely driven by soil pH and organic matter content.

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ABSTRACT

Information regarding the impact of low concentration of engineered nanoparticles on soil microbial communities is currently limited and the importance of soil characteristics is often neglected in ecological risk assessment. To evaluate the impact of TiO₂ nanoparticles (NPs) on soil microbial communities (measured on bacterial abundance and carbon mineralization activity), 6 agricultural soils exhibiting contrasted textures and organic matter contents were exposed for 90 days to a low environmentally relevant concentration or to an accidental spiking of TiO₂-NPs (1 and 500 mg kg⁻¹ dry soil, respectively) in microcosms. In most soils, TiO₂-NPs did not impact the activity and abundance of microbial communities, except in the silty-clay soil (high OM) where C-mineralization was significantly lowered, even with the low NPs concentration. Our results suggest that TiO₂-NPs toxicity does not depend on soil texture but likely on pH and OM content. We characterized TiO₂-NPs aggregation and zeta potential in soil solutions, in order to explain the difference of TiO₂-NPs effects on soil C-mineralization. Zeta potential and aggregation of TiO₂-NPs in the silty-clay (high OM) soil solution lead to a lower stability of TiO₂-NP-aggregates than in the other soils. Further experiments would be necessary to evaluate the relationship between TiO₂-NPs stability and toxicity in the soil.

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1. Introduction

Titanium dioxide nanoparticles (TiO₂-NPs) are engineered nanomaterials that are commonly used in diverse domains, such as cosmetics, sunscreens, paints, or coatings [1,2]. A large fraction of TiO₂-NPs in commercial products can reach natural ecosystems, especially the soil, which is one of the biggest sink for nanomaterials

[3–5]. The major pathways for TiO₂-NPs to enter soil are through agricultural amendments of sewage sludge or after an accidental spill during industrial production [6,7]. The increasing environmental release of TiO₂-NPs raises concerns about the potential effect on soil functioning and consequently on its capacity to fulfill essential ecosystem services.

Soils are porous systems consisting in complex structured assemblies of mineral and organic particles combined with liquid and gaseous phases. Soil properties such as clay and organic matter (OM) contents or pH greatly influence the behavior and bioavailability of common pollutants like pesticides, heavy metals or polycyclic aromatic hydrocarbons [8–12]. However, little

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information is available on the influence of soil properties on the aggregation and surface charge of NPs, which will determine their mobility and bioavailability [13–15].

Vittori Antisari et al. [16] demonstrated that metal oxide NPs are localized in small size (2–53 and <2 μm) soil aggregates indicating that such NPs interact essentially with clay minerals and OM. This preferential accumulation of NPs in the clay fraction suggests that NPs can be in contact with most of microbial communities inhabiting soil, since 40–70% of total soil bacteria are located in the 2–20 μm and <2 μm fractions [16,17].

Soil microorganisms are key players of many critical functions, such as biogeochemical cycling (e.g. carbon, nitrogen, sulphur and phosphorous cycles), plant productivity or climate regulation. Microbial communities are known to be sensitive ecological indicators of soil response to environmental perturbations [18–20] and can be good models to investigate TiO₂-NPs effect on soil functioning and quality [21]. Previous studies have shown that TiO₂-NPs can induce a decrease in microbial activity and a shift in bacterial community structure in soils [22–25]. However, these pioneer studies were performed only with high concentrations of TiO₂-NPs and usually with a single model soil. Since extrapolation of results from one contaminated soil to another is difficult because of the great heterogeneity of soils in terms of composition, structure and reactivity, studies conducted under more realistic concentrations and covering a wide range of soils are required.

In this study, we investigated TiO₂-NPs microbial ecotoxicity in 6 agricultural soils exhibiting contrasted texture (sandy-loam, loam and silty-clay) and organic matter content (low or high concentration). Since soil properties, such as texture or organic matter content, influence microbial community composition and the potential bioavailability of TiO₂-NPs, we expected contrasted TiO₂-NP toxicity in the different soils. We assessed the effects of TiO₂-NPs on soil microbial communities in a microcosm experiment, by measurements of carbon (C) mineralization activity and soil bacterial abundance using quantitative PCR. The soils were exposed to a low TiO₂-NPs concentration (1 mg kg⁻¹ dry soil) in the range of the predicted TiO₂-NPs concentrations in soil [5] and a higher concentration representing an accidental pollution (500 mg kg⁻¹ dry soil). In order to interpret and explain the differences of TiO₂-NPs effects on soil microbial communities, we characterized the apparent size and surface charge of TiO₂-NPs in soil solutions of the 6 agricultural soils using Dynamic Light Scattering (DLS).

2. Materials and methods

2.1. Soils

This study was conducted with soils belonging to three different textural classes: a sandy-loam, a loam and a silty-clay. Soils were collected from the upper 20 cm layer of 3 different agricultural sites located in the Rhône-Alpes and Burgundy regions

of France. At each site, the sampling was performed in 2 plots, distant of less than 500 m with low and high OM contents. After collection, visible rocks, roots and plant litter were manually removed. The soil was sieved (2 mm) and homogenized before storage at 4 °C. The sandy-loam soil (Cambisol, WRB, 2006) was obtained at the SERAIL (Station Expérimentale Rhône-Alpes et Information Légumière), an agricultural research station at Brindas (Rhône, France) from a field under vegetable rotation. The high OM content soil was collected in a plot, which had been fertilized with organic matter (bark compost) since 1995. The low OM content soil was collected in another plot, which received no amendment during the same period. The loamy soil (Luvisol, WRB, 2006) was collected at La Côte Saint-André (Isère, France) in a field cropped with maize (low OM content) and under permanent pasture (high OM content). The silty-clay soil (Cambisol, WRB, 2006) was sampled at Commarin (Côte d'Or, France) from a field in rape-wheat-winter barley rotation (low OM content) and under an adjacent permanent pasture (high OM content). Soils were characterized (Table 1) by the Laboratoire d'analyse des sols (LAS, Arras, France), for particle-size analysis (textural class), OM content, cation exchange capacity (CEC) and titanium content. pH and ionic strength were measured according to ISO 10390 and ISO 11265 procedures. Main soil elements (Table S1) were also measured following US-EPA 3052 guidelines, using a microwave assisted (Novawave, SCP Science) acid digestion (HF+HNO₃). Element concentrations were determined using ICP-OES (Varian 700-ES).

2.2. Soil solutions

Soil solutions were prepared to characterize TiO₂-NPs in physicochemical conditions as close as possible to those encountered in soil in terms of pH, ionic strength and dissolved components. The protocol was designed to obtain the dissolved components present in soils after removal of soil colloids larger than the initial size of TiO₂-NPs (i.e. 20 nm) from the suspension. Dynamic light scattering (DLS) analysis does not allow differentiating the introduced TiO₂-NPs from the native nanometric soil components. The removal of possibly interfering soil components is thus a crucial step in the reliable measurement of TiO₂ NPs. Soil solutions were prepared by shaking 10 g of soil dispersed in 50 mL of ultrapure water (18 MΩ) during 30 min at 180 rpm and 20 °C in a refrigerated incubator shaker (New Brunswick – Eppendorf, Hamburg, Germany). The solutions were then centrifuged for 20 min at 8000 × g, 20 °C (Centrifuge 5804R, Eppendorf, Hamburg, Germany) to eliminate particles larger than 20 nm according to the Stokes' law. The supernatants were collected and stored at 4 °C. For each soils, solutions were prepared in triplicates. pH, ionic strength, cation concentrations (Ca, Fe, Na) using microwave assisted acid digestion and determined by ICP-OES (Varian 700-ES) and dissolved organic carbon (DOC) using a Shimadzu TOC-V (Shimadzu, Kyoto, Japan) were assessed in each soil solution (Table 1).

Table 1
Main properties of soils and soil solutions.

	Soils									Soil solutions					
	% Sand	% Loam	% Clay	% OM	WHC (X)	pH	Ionic strength (mM)	CEC (cmol ⁺ kg ⁻¹)	Ti (g kg ⁻¹)	pH	Ionic strength (mM)	DOC (mg L ⁻¹)	Ca (mg L ⁻¹)	Fe (mg L ⁻¹)	Na (mg L ⁻¹)
Sandy-loam – Low OM	68.4	14.7	16.9	2.09	20	7	0.98	11.5	2.44	6.7	1	11.6	21.7	1.45	8.89
Sandy-loam – High OM	65.6	16.1	18.3	4.46	20	6.9	1.59	15.4	2.38	6.4	2	18.6	12.2	1.89	12.1
Loam – Low OM	37.5	42.7	19.8	2.23	30	6.4	0.60	8.79	2.71	6.2	1.2	8.2	28.8	1.90	13.2
Loam – High OM	40.3	40.8	18.9	6.77	30	6.3	1.31	15.3	2.4	5.4	2.4	10.5	38.2	2.79	21.8
Silty-clay – Low OM	8.2	49.8	42.0	4.72	47	6.9	0.51	17.4	5.26	6.3	1.5	25.2	33.5	1.79	11.4
Silty-clay – High OM	10.1	50.8	39.1	7.87	51	7.7	1.37	20.1	4.52	7.1	1.6	8.9	80.8	3.53	5.48

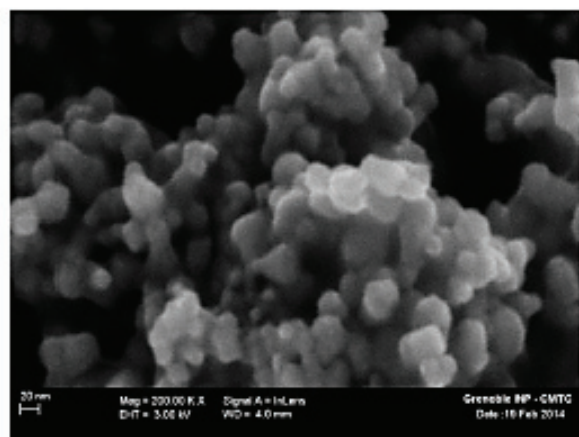


Fig. 1. Typical morphology of TiO₂ nanoparticles measured by SEM-FEG.

2.3. Nanoparticles characterization in ultrapure water and soil solutions

Titanium dioxide nanoparticles (TiO₂-NPs) were provided by Sigma-Aldrich (St Louis, USA) as a mixture of anatase (80%) and rutile (20%) crystal structure with at least 99.5% purity. According to the manufacturer information, TiO₂-NPs presented a specific surface area of 35–65 m² g⁻¹ and a mean particle size of 21 nm in powder as measured by Transmission Electron Microscopy. The particle size of TiO₂-NPs was verified (Fig. 1) using a ZEISS Ultra 55 scanning electron microscopy–field emission gun (SEM-FEG) and energy dispersive spectroscopy (EDS) with a SDD detector (BRUKER AXS-30 mm²). TiO₂-NPs were dispersed in ultrapure water and a droplet of the suspensions was deposited on an aluminum support and allowed to evaporate at room temperature. The samples were then metalized with carbon using thermal vaporization in a Gatan apparatus (PECS 682). The average particle size of TiO₂-NPs measured was 28.7 nm.

The hydrodynamic diameter and zeta potential of TiO₂-NPs in water or soil solutions were determined by dynamic light scattering (DLS) using a Zetasizer NanoZS (Malvern Instruments – UK). The surface charge was measured by electrophoretic mobility measurements converted to zeta potential and the particle size is presented as hydrodynamic diameter. TiO₂-NPs suspensions (100 mg L⁻¹) were prepared in ultrapure water or in soil solutions. All TiO₂-NPs suspensions were dispersed through ultrasonication for 5 min before use to ensure suspensions homogeneity. For each sample, the mean of 3 measurements was recorded. To determine the zero point of charge (ZPC) of TiO₂-NPs suspended in ultrapure water or in soil solutions, measurements of zeta potential were done at pH varying from 3 to 8 using an MPT-1 automatic titrator (Malvern Instrument) (HNO₃ (0.1 M) or NaOH (0.1 M) additions). For each measurement, control samples without TiO₂-NPs were analyzed to ensure that soil colloids did not interfere with the NPs analysis.

2.4. Soil microcosms and experimental design

One kg of soil (equivalent dry weight – eq dw) was spiked with a volume of TiO₂-NPs suspensions prepared in ultrapure water at 20 mg or 10 g TiO₂ L⁻¹ in order to obtain 1 mg TiO₂ kg⁻¹ or 500 mg TiO₂ kg⁻¹ dry soil respectively and to increase the soil water content to the water holding capacity (WHC). The exposure doses were chosen to represent a low realistic environmental concentration (1 mg kg⁻¹ dry soil) [5] and a high concentration representing

an accidental spill (500 mg kg⁻¹ dry soil). The TiO₂-NPs suspensions were added homogeneously using a multichannel pipette and then soils were thoroughly mixed for 10 min to ensure a uniform spiking. Soils receiving only the same volume of ultrapure water were used as controls. Microcosms were set up by placing 50 g of control or spiked soils (eq dw) into 150 mL glass plasma flasks sealed with rubber stoppers to avoid soil drying and to maintain the soil moisture constant during the experiment. Microcosms were incubated for 0, 7, 30 and 90 days in the dark at 28 °C. Microcosms were weekly aerated for 5 min to ensure a renewal of the atmosphere of the flasks. This experimental design resulted in a total of 324 microcosms with 6 replicates per treatment (6 soils × 3 concentrations of TiO₂-NPs × 3 sampling times × 6 replicates). At the beginning of the experiment, 36 additional microcosms (6 soils × 6 replicates) were prepared to measure the baseline conditions (T = 0 day) of the soil samples.

At the end of each incubation time, microcosms were subsampled as follows: 10 g of soil (eq dw) were immediately used for measurements of C-mineralization, 3 g of soil was stored at –20 °C before DNA extraction. The remaining samples were kept at 4 °C for further soil physicochemical analysis. Soil pH was measured after incubation with NPs in every soil microcosms according to the procedure cited above. Soil pH was not affected by the addition of TiO₂-NPs at the 3 sampling times (data not shown).

2.5. C-mineralization

Substrate induced respiration (SIR) rate, i.e. C-mineralization potential was measured according to Patra et al. [26]. Fresh soil collected immediately after sampling (10 g eq dw) was placed in a 150 mL plasma flask. Distilled water (0.5 mL) containing glucose was added to achieve a final concentration of 1.2 mg C-glucose g⁻¹ dry soil. The flasks were sealed with rubber stoppers and incubated at 28 °C for 6 h. After 2 h of initial incubation, gas samples were analyzed every hour during the incubation to measure the increase in CO₂ emission using a gas chromatograph (Micro GC R3000, SRA Instrument, Marcy L'Etoile, France). C-mineralization was expressed as μg C-CO₂ h⁻¹ g⁻¹ dry soil.

2.6. DNA extraction and quantification

DNA was extracted from 0.5 g of frozen soil using the Power Soil™ DNA Isolation Kit (MO BIO laboratories, Carlsbad, CA, USA), following the manufacturer's instructions and then quantified using the Quant it™ Picogreen® dsDNA Assay kit (Molecular Probes, USA). Fluorescence was measured with a UV spectrophotometer Xenius (Safas, Monaco) (λ = 520 nm).

2.7. Quantification of the abundance of soil total bacteria

The abundance of soil total bacteria was measured by quantitative PCR targeting universal gene *rps* coding for 16S rRNA. Amplification was performed using gene primers 519F and 907R [27,28].

The final reaction volume was 20 μL and contained (final concentrations): 0.3 μM of each primer, 1X of Lightcycler 480 Probes Master Mix (Roche Diagnostics, Meylan, France) and 5 ng of soil DNA extract or 10⁸–10² 16S copy number of the pQuantAlb plasmid [29]. The amplification program was 10 min at 95 °C followed by 40 cycles of 15 s at 95 °C, 1 min at 63 °C and 30 s at 72 °C and finally 10 s at 40 °C. Melting curves analysis confirmed the specificity of the amplification. All quantitative PCR reactions including unknown samples and standard curves were performed in duplicate.

2.8. Statistical analysis

All results are presented as means (\pm standard error). For each soil type, a one-way analysis of variance (ANOVA) and post hoc Tukey HSD were performed to test the effect of TiO_2 -NPs concentration on measured variables at each sampling time. When necessary, data were log-transformed prior to analysis to ensure conformity with the assumptions of normality and homogeneity of variances. We also conducted multiple regression to identify the drivers of C-mineralization in soils. The TiO_2 -NPs concentration and every soil parameters that were significantly related to C-mineralization were kept in the final model. Effects with $P < 0.05$ are referred to as significant. All statistical analysis was carried out using R statistical software 2.13.2 [30].

3. Results

3.1. Effect of TiO_2 -NPs on C-mineralization

C-mineralization was not affected by the addition of TiO_2 -NPs in most of the soils tested (sandy-loam – low and high OM; loam – high OM; silty-clay – low OM) at the 3 sampling dates (Fig. 2). However, after 7 days, a decrease of C-mineralization was observed in the loam soil (low OM) for the dose 1 (-22.7% , $P = 0.008$, Fig. 2A) but this effect appeared transitory and was no longer observed at 30 and 90 days of incubation. In the silty-clay soil (high OM) a significant decrease of C-mineralization over time was observed with both TiO_2 -NPs concentrations (Fig. 2).

However, no clear dose-response was noticed since both concentrations induced approximately the same decrease of C-mineralization after 7 days (dose 1: -19.8% , $P = 0.01$; dose 500: -21.2% , $P = 0.006$) and 30 days (dose 1: -14.2% , $P = 0.02$; dose 500: -12.1% , $P = 0.05$). After 90 days, a decrease was observed only for the dose 500 (-15.9% , $P = 0.001$), whereas a trend to resiliency occurred for the dose 1 as shown in Fig. 3.

3.2. Effect of TiO_2 -NPs on bacterial abundance

In most cases, the bacterial abundance was not significantly affected by the presence of TiO_2 -NPs (Table 2), except in the loamy soil (low OM) 7 days after spiking with 500 mg kg^{-1} (-25% , $P = 0.03$). In the silty-clay soil (high OM), decreases of 24% and 37% were observed after 90 days for the TiO_2 -NPs doses 1 and 500, respectively. However, these decreases were not significant because of high variability in the quantitative PCR data (Table 2).

3.3. TiO_2 -NPs characteristics in water and soil solutions

The average size of TiO_2 -NPs measured by DLS in ultrapure water was 158 nm (± 7.2) (Fig. 4). In control soil solutions (without TiO_2 -NPs), colloid sizes did not exceed 28.9 nm (± 3.3), consistently to what was expected according to the Stokes law (Fig. 4).

When TiO_2 -NPs were added to soil solutions, the peak corresponding to the size of the natural soil colloids (around 20 nm) was no longer detected, indicating that their relative abundance were negligible compared to that of added TiO_2 -NPs (100 mg L^{-1}) (data not shown). In soil solutions, TiO_2 -NPs were always significantly less aggregated (smaller apparent size) than in ultrapure water except in the solution of the silty-clay soil (high OM) (Fig. 4). The sizes of TiO_2 -NPs measured in the soil solutions were not significantly different except between the two silty-clay soil solutions, with size values ranging from 74 nm (± 14.9) in the silty-clay soil with low OM content to 128 nm (± 21.7) in the silty-clay soil with high OM content (Fig. 4). When dispersed in ultrapure water, TiO_2 -NPs presented a zeta potential of -13.4 mV (± 0.5) and a zero point

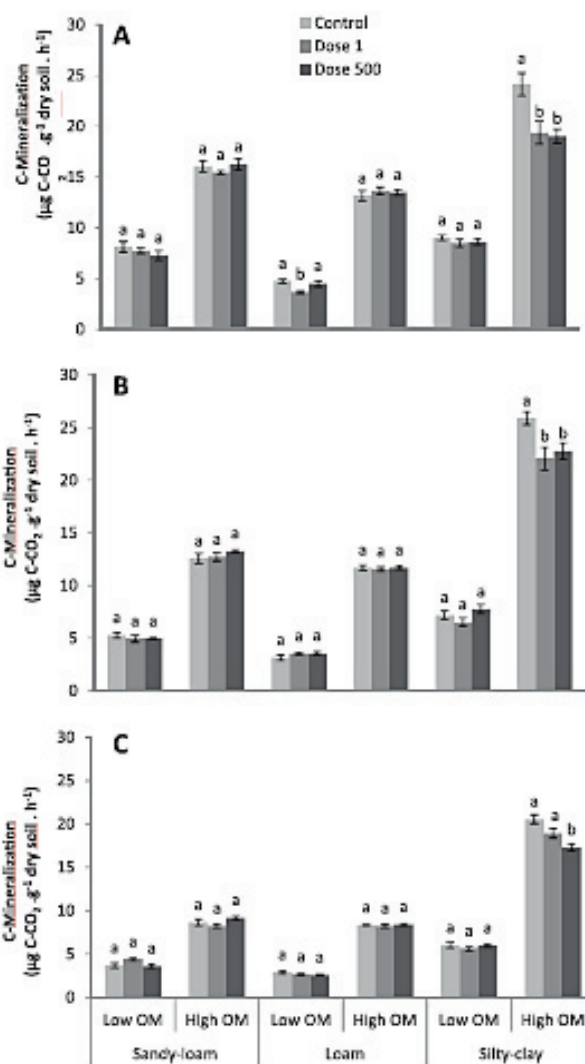


Fig. 2. C-mineralization in the 6 soils after (A) 7, (B) 30 and (C) 90 days in the different treatments (Control, Dose 1, Dose 500 mg kg^{-1} TiO_2 -NPs). Means and standard errors are presented ($n = 6$). For each soil, bars labeled with the same letter do not differ at $P < 0.05$.

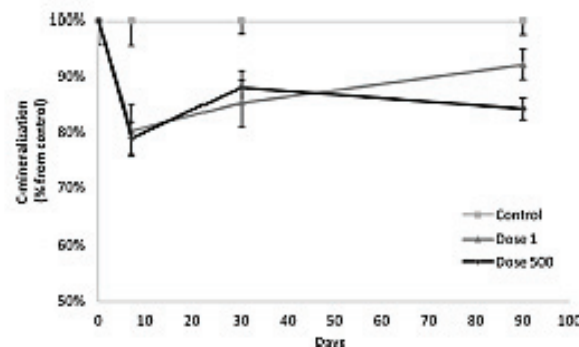


Fig. 3. Changes in C-mineralization in the silty-clay soil (high OM) in the different treatments (Control, Dose 1, Dose 500 mg kg^{-1} TiO_2 -NPs). Data are expressed as the percentage change from the control treatment. Means and standard errors are presented ($n = 6$).

Table 2
Bacterial abundance ($\times 10^9$ rrs copy number g^{-1} dry soil) in the different TiO_2 -NPs treatments (Control, Dose 1, Dose 500 $mg\ kg^{-1}$ TiO_2 -NPs) after 7, 30 and 90 days. Means \pm standard errors are presented ($n=6$).

	7 days			30 days			90 days		
	Control	Dose 1	Dose 500	Control	Dose 1	Dose 500	Control	Dose 1	Dose 500
Sandy-loam – Low OM	5.85 \pm 0.3	5.91 \pm 0.4	6.56 \pm 0.2	3.07 \pm 0.7	3.46 \pm 0.5	2.62 \pm 0.6	n.d.	n.d.	n.d.
Sandy-loam – High OM	5.14 \pm 0.3	5.09 \pm 0.5	6.51 \pm 0.5	4.08 \pm 0.6	4.02 \pm 0.3	3.64 \pm 0.4	n.d.	n.d.	n.d.
Loam – Low OM	5.77 \pm 0.4	4.84 \pm 0.3	4.32 \pm 0.3	3.3 \pm 1.1	3.3 \pm 1.2	2.82 \pm 0.7	n.d.	n.d.	n.d.
Loam – High OM	7.43 \pm 1.6	6.76 \pm 0.3	6.25 \pm 1.2	6.94 \pm 1.0	6.31 \pm 1.0	6.61 \pm 0.9	n.d.	n.d.	n.d.
Silty-clay – Low OM	7.69 \pm 1.6	5.38 \pm 1.3	7.08 \pm 1.1	5.30 \pm 1.8	3.44 \pm 1.0	5.30 \pm 1.5	n.d.	n.d.	n.d.
Silty-clay – High OM	38.2 \pm 5	32 \pm 7	31.7 \pm 4	33.1 \pm 4	30.4 \pm 5	31.5 \pm 3	29.9 \pm 4	22.8 \pm 5	18.3 \pm 2

Value in bold is significant ($P < 0.05$); n.d. is not determined.

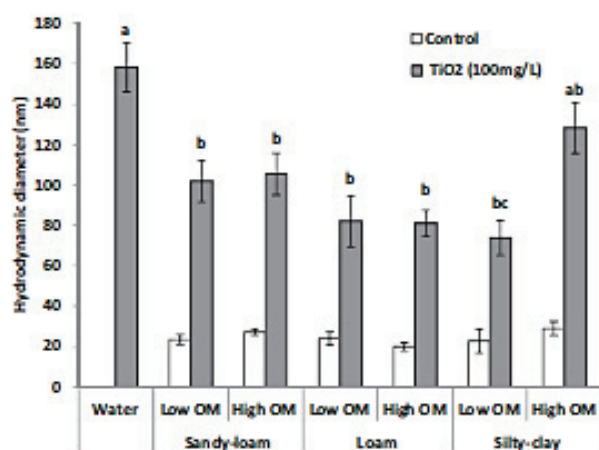


Fig. 4. Hydrodynamic diameters of TiO_2 -NPs ($100\ mg\ TiO_2\text{-NPs}\ L^{-1}$) measured in ultrapure water and soil solutions by dynamic light scattering (DLS). Controls are soils solutions without TiO_2 -NPs added. Means and standard errors are presented ($n=3$). Means followed by the same letter are not significantly different at $P < 0.05$.

of charge at pH 5.5 (Fig. 5). In contrast, when dispersed in soils solutions, TiO_2 -NPs were always negatively charged whatever the pH (Fig. 5).

4. Discussion

TiO_2 -NPs can be toxic toward bacteria through membrane disorganization and reactive oxygen species (ROS) production due

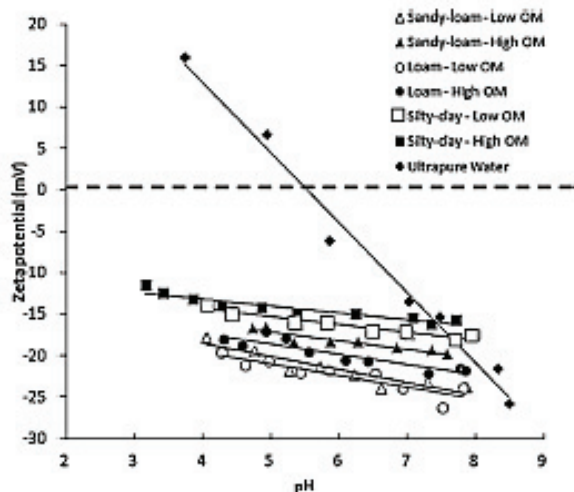


Fig. 5. Zeta potential of TiO_2 -NPs in function of pH measured in ultrapure water and soil solutions by DLS.

to their adsorption to cell membrane and photocatalytic oxidation [31,32]. However, in most soils of the present study, TiO_2 -NPs had no significant effect on either C-mineralization potential activity or on bacterial abundance, whatever the concentration (1 or 500 $mg\ kg^{-1}$ dry soil) over 90 days. The C-mineralization and bacterial abundance values were in the range of other studies carried out with arable or grassland soils [23,26,33,34]. A decrease of C-mineralization with 500 $mg\ kg^{-1}$ of TiO_2 -NPs over 60 days has been reported in a loam soil [23]. In the present study, in both loam soils tested, the same concentration had no effect, except a slight and transitory decrease of bacterial abundance after 7 days. Using another metal NP, Pawlett et al. [35] reported that the impact of zero-valent iron NPs on C-mineralization was highly soil-dependant. Only the silty-clay soil (high OM) was strongly affected by TiO_2 -NPs contamination. The absence of negative effects in the silty-clay (low OM) soil suggests that soil clay content does not influence TiO_2 -NPs toxicity. Conversely, TiO_2 -NPs toxicity may be related to soil OM content and pH, as the negative effects on C-mineralization were only observed in the soil exhibiting the highest pH and OM content. These assumptions are supported by a multiple regression analysis (Table S2) that highlights a significant interaction between TiO_2 -NPs effects and pH or OM content, but not with clay content. Further soil experiments specifically designed to test a wider range of OM content and pH are needed to generalize the role of these parameters in TiO_2 -NPs toxicity.

The toxicity of NPs is known to be influenced by their aggregation state and stability [32,36], which are controlled by environmental factors. The role of OM as a dispersing and stabilizing agent of TiO_2 -NPs in aqueous media has been reported [37,38]. Consistently, TiO_2 -NPs appeared more dispersed in soil solutions than in ultrapure water in the present work. This could be due to the interactions of dissolved OM and NPs that are known to change the NPs surface properties and/or increase steric repulsions [39,40]. The aggregation of TiO_2 -NPs was measured and compared between the different soil solutions exhibiting contrasted physicochemical properties. No significant differences were observed, except in the silty-clay soil with low OM and high OM content. In the silty-clay (high OM) soil solution, TiO_2 -NPs were the most aggregated. In the latter case, the size of NPs aggregates was not significantly different from the size measured in ultrapure water. The low concentration of DOC in this soil solution could explain the similarity of TiO_2 -NPs aggregation in this soil and in pure water. Another explanation could be the high concentration of Ca in relation with the alkaline pH of this soil. Indeed it has been demonstrated that aggregation of NPs increases in presence of divalent cations [37,41].

The ionic strength is also a key environmental factor for NPs properties. As observed in this study, it is normally less than 0.005 M in soil solution [42]. Consistently with the literature, the ionic strength of soil solutions was positively correlated with the zeta potential of TiO_2 -NPs ($R^2 = 0.50$, $P = 0.01$, $n = 18$) [43,44]. The changes of NPs zeta potential with ionic strength are the result of

electrostatic double layer compression by charge screening [43]. As previously observed by Zhang et al. [44], the zeta potential of TiO₂-NPs measured in the 6 soils solutions was always negative in presence of natural OM, within the tested pH range. At the opposite, in ultrapure water TiO₂-NPs presented a ZPC at pH=5.5, which is consistent with the literature [40,45,46]. The negative zeta potential of TiO₂-NPs in soil solutions may be due to the imparting of negative charges from OM to NPs surfaces [44]. Moreover a positive correlation between zeta potential and iron was observed ($R^2=0.43$, $P=0.02$, $n=18$), emphasizing the potential key role of cations on NPs properties. The increase of zeta potential with cation concentration has been explained by the neutralization of the negative charge that OM imparted to NPs [44].

The strongest effects of TiO₂-NPs on C-mineralization were observed in the silty-clay soil (high OM). In the corresponding soil solution, TiO₂-NPs were less negatively charged and more aggregated than in the others, due to the high ionic strength, the high Ca concentration and the low DOC content. Fang et al. [47] found a positive correlation between TiO₂-NPs stability in soil suspensions with DOC content and a negative correlation was observed with ionic strength, pH and zeta potential. Based on these results, it is likely that TiO₂-NPs in the silty-clay (high OM) soil solution exhibit properties conferring low stability in suspension. More research are needed to determine whether the impact of TiO₂-NPs on microbial activity in the silty-clay soil (high OM) can be related to the instability of TiO₂-NPs aggregates or not. Microscale studies of the interactions between microbes, NPs and soil components would be very useful to evaluate the relationship between the TiO₂-NPs stability and toxicity in soil.

Besides the role of the physicochemical NPs properties, the absence of TiO₂-NPs effect on C-mineralization in most of the soils could also depend on the composition of the indigenous microbial community, which may be more resistant to such NPs perturbation than the silty-clay soil with high OM content. Future research should address the role of soil microbial diversity in the response to TiO₂-NPs toxicity to clarify the impact on soil functioning.

Only few studies have investigated the effect of low concentrations of NPs on soil microbial communities [16,49,50]. To our knowledge, this is the first work evaluating the impact of such a low concentration of TiO₂-NPs on soil microorganisms. By using two doses of NPs, we expected to observe a dose-response of TiO₂-NPs that would lead to a higher decrease of C-mineralization with 500 mg kg⁻¹ than with 1 mg kg⁻¹. This was not observed here, as the low concentration induced similar or even higher negative effects than a 500-fold higher concentration (Fig. 2A and B). The absence of a clear dose-response on C-mineralization has already been reported by Ge et al. [23] for 0.5–2 g of TiO₂-NPs kg⁻¹. These results are likely due to NPs homo- and hetero-aggregation processes, which are controlled by the NPs concentrations applied to soil [48], explaining the absence of linear dose-response observed in soil.

5. Conclusion

This work shows that a low concentration of TiO₂-NPs can have detrimental effects on microbial mineralization in soil. However, contrasted responses to TiO₂-NPs spiking were observed among the six studied soils. Contrary to what is observed with common pollutants (such as heavy metals or polycyclic aromatic hydrocarbon) that cause more toxic effect in coarse-textured soils [8,51], TiO₂-NPs induced a lasting decrease of microbial activity only in a silty-clay soil, whereas sandy soils were not affected. It seems that soil texture is not a crucial factor in TiO₂-NPs toxicity. Among the multiple environmental factors, pH and OM content might be the major drivers that affect NPs toxicity in soil. Although not reflecting exactly the physicochemical conditions encountered in soil, the

characterization of TiO₂-NPs properties in soil solutions allowed to propose hypotheses explaining these results. We assume that the properties of TiO₂-NPs measured in the silty-clay (high OM) soil solution lead to a lower stability of NPs aggregates that might explain their higher toxicity to microorganisms than in the other soils.

Altogether, our results showed that the use of low concentrations of NPs coupled with their characterization directly in solutions of a wide range of natural soils provided useful information for ecotoxicological studies and appeared to be a relevant step toward a better understanding of the impact of NPs on soil functioning.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.jhazmat.2014.10.004>.

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SUPPLEMENTARY INFORMATION

Table S1 Concentration of the main elements in the 6 soils

Concentrations in mg kg ⁻¹	Ag	Al	B	Ca	Cd	Co	Cr	Cu	Fe	K	Mg	Mn	Na	Ni	Pb	Sr	Zn
Sandy-Loam (Low OM)	0.46	10888	16.01	1195	1.80	6.05	32.53	17.06	11867	6430.3	143.73	290.43	1445.9	15.55	33.26	18.85	47.52
Sandy-Loam (High OM)	0.45	10357	15.84	1189	1.25	5.43	27.80	21.70	8352	7668.7	151.58	274.41	1423.4	11.00	32.21	18.32	50.66
Loam (Low OM)	0.40	12011	27.13	1868	1.25	5.82	37.65	10.71	10159	6762.8	867.25	545.35	2203.6	13.48	38.60	42.27	47.24
Loam (High OM)	0.66	5509	65.95	3499	1.65	8.67	57.93	26.51	12154	7771.2	185.28	671.21	2557.5	24.93	48.22	34.25	84.11
Silty-Clay (Low OM)	0.77	4835	83.81	825.8	2.49	18.50	83.38	18.78	18108	8920.1	51.44	768.08	1605.5	37.89	32.40	18.96	112.28
Silty-Clay (High OM)	0.69	9314	74.62	3026	2.22	15.18	66.88	20.49	16064	8725.3	570.44	618.65	1333.7	37.57	31.68	17.89	117.00

Table S2 Relationship between C-mineralization and TiO₂-NPs and soil variables tested by multiple regression to identify the main drivers of C-mineralization in soils and to determine which soil physicochemical properties influence TiO₂-NPs toxicity. The following variables were included in the model: TiO₂-NPs treatments, cation exchange capacity (CEC), ionic strength, soil pH, OM content and clay content. To determine if TiO₂-NPs toxicity was related to soil physicochemical properties, the interactions between TiO₂-NPs and the different soil variables were tested. Analysis was performed on the complete set of data, including the 6 soils and the 3 sampling times (n=324). Values in bold are significant.

Variables	P-value
TiO ₂	0.02 *
CEC	< 0.001 ***
Ionic Strength	< 0.001 ***
pH	< 0.001 ***
OM content	< 0.001 ***
Clay content	< 0.001 ***
TiO ₂ x CEC	0.56
TiO ₂ x Ionic Strength	0.46
TiO ₂ x pH	0.03 *
TiO ₂ x OM content	0.04 *
TiO ₂ x Clay content	0.58
Multiple regression model	P-value < 0.001 ***
Model R ²	0.87

4. Conclusions du chapitre

Dans ce chapitre, le but des expérimentations était de déterminer les propriétés clés du sol qui conditionnent le devenir et la toxicité des NPs d'oxydes métalliques. Nous avons pu mettre en évidence que quel que soit le sol considéré, les NPs de TiO_2 et de CuO sont très peu mobiles et que leur transport pouvait être affecté par la matière organique dissoute présente dans la solution du sol, en accord avec divers travaux menés dans des milieux simplifiés. Cependant, la teneur en argile n'apparaît pas comme un paramètre clé influençant le transport des NPs étudiées.

La faible mobilité observée des NPs suggère qu'elles sont rapidement filtrées et retenues dans les premiers centimètres du sol. Le risque de transport de ces contaminants hors du site de pollution initial semble donc faible. Ces résultats encouragent à évaluer quelle est la toxicité de ces composés à la suite de leur introduction dans les sols sur le long terme.

De manière générale, nos résultats indiquent que les TiO_2 -NPs ont un faible potentiel toxique vis-à-vis de la communauté microbienne hétérotrophe dans les sols étudiés et ce, quelle que soit la concentration appliquée. Dans le sol limono-argileux où une diminution significative de la minéralisation du carbone a été observée, nous n'avons pas observé d'effet dose-réponse contrairement à ce qui était attendu. L'utilisation de six sols aux propriétés contrastées nous a permis d'identifier le pH et la MO comme des paramètres potentiellement impliqués dans la toxicité des NPs. Ceci nécessiterait cependant d'être approfondi en s'intéressant plus particulièrement au rôle de la nature et de la concentration en carbone organique dissout. Par ailleurs, la généralisation de nos conclusions passe par la confirmation des observations à partir d'un panel plus large de sols présentant des propriétés physico-chimiques contrastées.

Dans cette partie du travail, la toxicité des NPs a été abordée en considérant les effets au niveau d'un processus microbien global reposant sur une communauté diversifiée présentant une forte redondance fonctionnelle. Dans la mesure où des processus clés du fonctionnement du sol sont assurés par des groupes fonctionnels moins diversifiés et présentant une redondance moins importante (cf. chapitre 1 § 6-d), nous avons choisi d'évaluer également les effets des NPs en ciblant les communautés nitrifiante et dénitrifiante.

Chapitre 2 : Influence des propriétés du sol sur le transport et la toxicité des nanoparticules d'oxydes métalliques

Par ailleurs, l'absence de relation dose-réponse observée étant surprenante, nous avons entrepris de vérifier l'absence de relation linéaire entre la concentration appliquée et la toxicité observée des TiO₂-NPs dans les sols en utilisant une gamme de concentrations en TiO₂ plus large. Le sol limoneux-argileux étant le seul dans lequel un effet toxique des NPs avait été observé, il a été retenu comme sol modèle pour la suite des investigations de ce travail.

Chapitre 3 : Impact du TiO_2 sur le cycle de l'azote : exemple d'un sol limoneux-argileux

1. Introduction

Dans le chapitre 2, nous avons montré que le cycle du carbone pouvait être affecté par les TiO₂-NPs dans un sol limono-argileux. Les processus microbiens du cycle de l'azote sont également cruciaux pour le fonctionnement biologique des sols. C'est pourquoi nous avons évalué les conséquences de la présence de ce polluant émergent sur les processus microbiens de nitrification et dénitrification. La nitrification est un processus souvent étudié en écotoxicologie car il est très sensible à la présence de polluants et est donc un indicateur sensible de perturbation des sols et de leur qualité.

Dans ce chapitre, nous nous sommes intéressés à l'impact de contaminations aiguë (Article 4 et 5) et chronique (Article 6) aux TiO₂-NPs sur la nitrification et la dénitrification dans un sol limono-argileux à fort taux de MO. Les effets ont été évalués sur les activités, les abondances et la diversité des groupes fonctionnels qui sont des indicateurs complémentaires pour décrypter l'impact d'une perturbation sur ces processus microbiens. Une approche originale en écotoxicologie par path analysis, permettant d'étudier les relations causales entre les différentes variables mesurées, a été utilisée en complément afin de comprendre comment la nitrification et la dénitrification étaient affectées par ce type de contaminant.

2. Effet d'une contamination aiguë sur le cycle de l'azote d'un sol

a. Article 4 : Présentation générale de l'étude et synthèse des principaux résultats

Dans le chapitre précédent, nous avons constaté que la minéralisation du carbone du sol limono-argileux (forte MO) était réduite d'environ 20 % quelle que soit la concentration de TiO₂-NPs. Dans la même expérimentation en microcosmes, nous avons donc aussi évalué les effets du TiO₂ sur la nitrification et la dénitrification. La nitrification est considérée comme étant un des processus microbiens les plus sensibles aux polluants alors que la dénitrification reste généralement peu affectée par ce type de perturbation (Bisset *et al.*, 2013). Toutefois, aucune étude n'est encore disponible sur l'évaluation de l'impact des TiO₂-NPs sur ces 2 étapes clés du cycle de l'N dans les sols.

Chapitre 3 : Impact du TiO₂ sur le cycle de l'azote : exemple d'un sol limoneux-argileux

Dans cette partie, nous avons évalué les conséquences des TiO₂-NPs sur les activités nitrifiantes et dénitrifiantes potentielles (NEA et DEA), ainsi que sur l'abondance des groupes fonctionnels impliqués (AOA, AOB et dénitrifiants portant les gènes *nirK* et *nirS*) par PCR quantitative (qPCR). Nous avons également étudié par séquençage haut-débit MiSeq (Illumina), l'impact du TiO₂ sur la diversité des communautés bactériennes et archées totales et impliquées dans l'oxydation de l'ammonium (AOB et AOA).

Nous avons observé que les activités nitrifiantes et dénitrifiantes ont été réduites en présence des 2 concentrations de TiO₂-NPs testées. La nitrification a été particulièrement affectée après 90 jours avec une diminution de 40%, tout comme les abondances des AOA et des AOB qui ont perdu 60 et 40%, respectivement, aux deux concentrations testées, sauf pour les AOB qui ne sont impactées qu'à 500 mg de NPs kg⁻¹ de sol. Aucun effet n'a été observé sur l'abondance des dénitrifiants. La diversité des archées, évaluée en ciblant les gènes *16S ADNr* et *amoA*, s'est révélée très faible avec la dominance très marquée d'un OTU (Operational Taxonomic Unit) du genre *Nitrososphaera* qui est une AOA appartenant au phylum des *Thaumarcheota*. L'abondance de cet OTU a été fortement réduite par les 2 doses de TiO₂-NPs et était bien corrélée à la nitrification et à l'abondance des AOA. L'évaluation de l'ensemble de ces résultats par path analysis suggère que les effets négatifs du TiO₂ sur l'OTU dominant *Nitrososphaera* expliquent en grande partie la diminution de l'activité nitrifiante observée qui a ensuite entraîné une diminution en cascade de l'activité dénitrifiante.

Nos résultats indiquent donc que dans le sol étudié la nitrification est un processus sensible aux TiO₂-NPs qui repose en grande partie sur l'activité des AOA *Nitrososphaera*. Ces dernières présentent une très faible diversité et donc une faible redondance fonctionnelle, ce qui a pour conséquence une faible résistance vis-à-vis des NPs. Comme attendu, la dénitrification a été moins affectée. Cependant, le fort couplage entre nitrification et dénitrification a conduit à une diminution de cette activité même si l'abondance des dénitrifiants est restée inchangée.

b. « Response of soil microbial community to titanium dioxide nanoparticles: a cascading pitch on the nitrogen cycle »

L'article 4 intitulé «Response of soil microbial community to titanium dioxide nanoparticles: a cascading pitch on nitrogen cycle» est en préparation pour une soumission dans *The ISME Journal*.

Response of soil microbial community to titanium dioxide nanoparticles: a cascading pitch on the nitrogen cycle

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Running title: Effects of TiO₂ nanoparticles on the N cycle

ABSTRACT

Soils are facing new environmental stressors, such as titanium dioxide nanoparticles (TiO₂-NPs). While these emerging pollutants are increasingly released into most ecosystems, including agricultural fields, their potential impact on soil and its functioning remain to be investigated. Here we report the response of the microbial community of an agricultural soil (silty-clay texture) exposed over 90 days to TiO₂-NPs at 1 and 500 mg kg⁻¹ dry soil. To assess their impact on soil functioning, we focused on the nitrogen cycle and thus measured nitrification and denitrification enzymatic activities combined with the quantification of specific representative genes (*amoA* for ammonia-oxidizers, *nirK* and *nirS* for denitrifiers). In addition, the bacterial, archaeal and ammonia-oxidizers diversity changes were investigated. With strong negative impacts on nitrification enzymatic activities and on the abundances of ammonia-oxidizing microbes, TiO₂-NPs triggered a cascading negative effect on denitrification activity and a deep modification of the bacterial community structure after 90 days of exposure even at the lowest and realistic NPs concentrations. These results appeal further research to assess how these emerging pollutants modify the soil ecosystem functioning and the related soil fertility.

INTRODUCTION

Titanium dioxide nanoparticles (TiO₂-NPs) are widely used into day life products such as sunscreens and toothpastes, industrial products like paints, lacquers and papers, as well as during photocatalytic processes such as water treatment (Keller *et al.*, 2013; Mitrano *et al.*, 2015). Consequently, TiO₂-NPs are chronically released into the environment either directly as nanofertilizers or nanopesticides (Servin *et al.*, 2015) or indirectly in agricultural soils through sewage-sludge application as fertilizers (Brar *et al.*, 2010). Despite their recognized importance in soil ecosystem functioning, the current literature lacks thorough investigations of the effect of TiO₂-NPs on soil microbial communities (Simonin and Richaume, 2015).

In soils, microbial communities play key roles in plant productivity, climate regulation or in biogeochemical processes (Falkowski *et al.*, 2008; Schimel and Schaeffer, 2012; Vacheron *et al.*, 2013), such as the nitrogen (N) cycle in which nitrification and denitrification processes control soil inorganic N availability and subsequent soil fertility (Philippot *et al.*, 2013). The strong coupling between these two functions makes the N cycle a good model to study direct and indirect impacts of environmental disturbances. Nitrification is a two-step process oxidizing ammonia into nitrate. It is assumed that the rate-limiting in nitrification is the oxidation of ammonia into nitrite (Kowalchuk and Stephen, 2001), performed by both ammonia-oxidizing bacteria (AOB) and ammonia-oxidizing archaea (AOA) (Prosser and Nicol, 2012). Nitrification is carried out by a group of microorganisms exhibiting a low functional diversity and is one of the most sensitive soil microbial processes to perturbations, such as pollutants contamination (Dalzell *et al.*, 2002; Broos *et al.*, 2005). However, the influence of environmental stressors on AOA is poorly documented (Mertens *et al.*, 2009; Ollivier *et al.*, 2012) because of the recent discovery of their involvement in soil nitrification (Leininger *et al.*, 2006).

Nitrification is followed by denitrification, which is the sequential reduction of oxidized N compounds (NO₃⁻, NO₂⁻) into gaseous products (NO, N₂O and N₂) (Zumft, 1997). Contrarily to nitrifier community, the ability to perform denitrification is widespread among several phylogenetic groups and this functional guild has been shown to be fairly insensitive to many

toxicants (Bissett *et al.*, 2013). However, the sensitivity of denitrifiers to TiO₂-NPs has not been investigated yet. Moreover, despite a potential resistance of the denitrifiers to this pollutant, a failure of nitrification may still result in a decrease of denitrification.

High concentrations of TiO₂-NPs have been showed previously to partly reduce general processes performed by many members of the microbial community such as microbial respiration and soil enzyme activities (Du *et al.*, 2011; Ge *et al.*, 2011; Simonin *et al.*, 2015). However, the impact of TiO₂-NPs at environmentally relevant concentrations on key processes performed by functional communities with different levels of functional redundancy, such as those involved in the N cycle, has never been investigated to date.

Here we report the effects of TiO₂-NPs on the N cycle in an agricultural soil exposed over 90 days to two concentrations of TiO₂-NPs simulating environmentally realistic contamination (1 mg kg⁻¹ dry soil) or an accidental pollution (500 mg kg⁻¹ dry soil) (Sun *et al.*, 2014). Microbial activities and abundances were assessed by measuring nitrification and denitrification enzymatic activities (NEA and DEA, respectively) combined to the quantification of specific representative genes (*amoA* for ammonia-oxidizers and *nirK* and *nirS* for denitrifiers). In addition, the effects of TiO₂-NPs on the diversity of *16S rDNA* bacterial and archaeal genes and *amoA* AOA and AOB genes were determined by high throughput sequencing. The consequences of TiO₂-NPs on the N cycle were discussed in the light of the information provided by a path analysis integrating different variables measured in the experiment.

MATERIAL AND METHODS

Nanoparticles, soil and experimental design

The TiO₂-NPs were provided by Sigma Aldrich (St Louis, USA) with a particle size of 21 nm in powder and ≥ 99.5 % purity. The size and surface charge were previously characterized by Dynamic Light Scattering (DLS) (Simonin *et al.*, 2015). The average size and zeta potential of TiO₂-NPs in ultrapure water used to spike the soil were 160 nm ± 7.2 and - 13.4 mV ± 0.5, respectively.

Chapitre 3 : Impact du TiO₂ sur le cycle de l'azote : exemple d'un sol limoneux-argileux

The soil was sampled from the upper 20 cm layer of a permanent pasture at Commarin (Côte d'Or, France), sieved (2 mm) and stored at 4°C before use. It is a silty-clay soil containing 39.1 % clay, 50.8 % loam and 10.1 % sand. The organic matter content was 7.87 %, the CEC 20.1 cmol⁺ kg⁻¹ and the water holding capacity 51 %. The soil pH (7.7) and the ionic strength (1.37 mM) were measured following ISO 10390 and ISO 11265 procedures, respectively. Soil pH was not modified by the addition of TiO₂-NPs throughout the experiment.

The soil was exposed to either a dose of 1 mg or 500 mg kg⁻¹ dry soil of TiO₂-NPs suspended in ultrapure water and dispersed in an ultrasonic bath (Bioblock Scientific) for 5 minutes just before use. TiO₂-NPs suspensions were added homogeneously using a multichannel pipette in order to achieve 100 % of the water holding capacity. The soil was thoroughly mixed manually for 10 minutes to ensure a homogeneous distribution of NPs. Control soils received only ultrapure water. Fifty grams (equivalent dry weight) of contaminated soils were placed into microcosms (150 mL plasma flask) sealed with rubber stoppers. Soil microcosms were incubated for 7, 30 or 90 days at 28°C in the dark and were weekly aerated to ensure a renewal of the atmosphere in the flasks. The experimental design resulted in 54 microcosms: 3 concentrations of TiO₂-NPs (0, 1, 500 mg kg⁻¹ dry soil) x 3 exposition times x 6 replicates per treatment. At the beginning of the experiment, 6 additional microcosms were prepared to assess the baseline conditions of the soil. For each incubation time, soil subsamples were immediately used for NEA and DEA measurements, and 3 g were stored at -20°C until DNA extraction.

Nitrification Enzymatic Activity (NEA)

NEA was determined following the protocol described in Dassonville *et al.* (2011). Briefly subsamples of fresh soil (3 g dry soil) were incubated with 6 ml of a solution of (NH₄)₂SO₄ (50 µg N-NH₄⁺ g⁻¹ dry soil). Distilled water was adjusted in each sample to achieve 24 ml of total liquid volume in flasks. The flasks were sealed with Parafilm[®] and incubated at 28°C under constant agitation (180 rpm). 1.5 ml of soil slurry were regularly sampled after 2h, 4h, 6h, 8h and 10h of incubation, filtered on 0.2 µm pore size and stored in vials at -20°C until measurement of NO₃⁻ concentrations using an ionic chromatography (DX120, Dionex, Salt

Lake City, USA) equipped with a 4×250 mm column (IonPac AS9 HC). NEA was expressed as $\mu\text{g N-NO}_3^- \text{ h}^{-1} \text{ g}^{-1}$ dry soil.

Denitrification Enzymatic Activity (DEA)

Potential denitrification was measured according to Patra *et al.* (2005). Briefly distilled water (1 mL) containing KNO₃ (50 $\mu\text{g N-NO}_3^- \text{ g}^{-1}$ dry soil), glucose (500 $\mu\text{g C-glucose g}^{-1}$ dry soil) and glutamic acid (500 $\mu\text{g C-glutamic acid g}^{-1}$ dry soil) were added to fresh soil (10 g equivalent dry soil) placed in a 150 ml plasma flask. The atmosphere was replaced by 90% helium to ensure anaerobic conditions and 10 % C₂H₂ was added to inhibit N₂O reductase activity. The flasks were sealed with rubber stoppers and incubated at 28°C for 6 h. After 2h of incubation, gas samples were analyzed every hour for the remaining incubation (5 hours) to measure N₂O concentration using a gas chromatograph (Micro GC R3000, SRA Instrument, Marcy L'Etoile, France). DEA was expressed as $\mu\text{g N-N}_2\text{O h}^{-1} \text{ g}^{-1}$ dry soil.

DNA extraction and quantification

DNA was extracted from 0.5 g of frozen soil using the Power Soil™ DNA Isolation Kit (MO BIO laboratories, Carlsbad, CA, USA) following the manufacturer's instructions and then quantified using the Quant it™ Picogreen® dsDNA Assay kit (Molecular Probes, USA). Fluorescence was measured using a UV spectrophotometer Xenius (Safas, Monaco) ($\lambda=520$ nm).

Abundance of nitrifying bacteria (AOA and AOB)

The abundance of AOA and AOB was measured by quantitative PCR targeting *amoA* gene sequences encoding for an ammonia monooxygenase specific for each group. Amplification was performed using gene primers *amoA_1F* and *amoA_2R* for the AOB (Rotthauwe *et al.*, 1997) and for the AOA (Tourna *et al.*, 2008). The final reaction volume was 20 μL and contained (final concentrations) 0.5 μM of each primer for the bacterial *amoA* or 0.75 μM of *CrenamoA616r* and 1 μM of *CrenamoA23f* for the archaeal *amoA*, 2 % bovine serum albumin

(BSA), 1X of QuantiTect SybrGreen PCR Master Mix (Qiagen, Courtaboeuf, France) and 10 ng of soil DNA extract or of DNA standards with 10^2 to 10^7 gene copies / μL using a constructed linearized plasmid containing archaeal (54d9 fosmid fragment, (Treusch *et al.*, 2005)) and bacterial (*Nitrosomonas europaea*, GenBank accession number:L08050) *amoA* genes. The samples were run in duplicate on a Lightcycler 480 (Roche Diagnostics, Meylan, France) as follows: for bacterial *amoA* 15 min at 95°C, 45 amplification cycles (30 s at 95°C, 45 s at 54°C, 45 s at 72°C and 15 s at 80°C) and 30 s at 40°C ; for archaeal *amoA*, 15 min at 95°C, 50 amplification cycles (45 s at 94°C, 45 s at 55°C and 45 s at 72°C) and 10 s at 40°C. Melting curves analysis confirmed the amplification specificity.

Denitrifying bacteria abundances

The abundance of denitrifying bacteria was measured by quantitative PCR of the genes *nirK* and *nirS* respectively encoding copper- and cytochrome cd1-containing nitrite reductase. Amplification was performed using *nirK876* and *nirK1040* gene primers (Henry *et al.*, 2004) or *nirSCd3aF* and *nirSR3cd* (Kandeler *et al.*, 2006). The final reaction volume for *nirK* quantification was 20 μL and contained (final concentrations) 1 μM of each primer, 0.02 μg of T4 gene protein 32 (QBiogene, Illkirch, France), 1X of QuantiTect SybrGreen PCR Master Mix (Qiagen, Courtaboeuf, France) and 5 ng of soil DNA extract or of standards with 10^2 to 10^7 copies of DNA (*Sinorhizobium meliloti* 1021). The samples were run in duplicate on a Lightcycler 480 (Roche Diagnostics, Meylan, France) as follows: 15 min at 95°C, 45 amplification cycles (15 s at 95°C, 30 s at 63°C and 30 s at 72°C) and 10 s at 40°C. The final reaction volume for *nirS* quantification was 25 μL and contained (final concentrations) 0.5 μM of each primer, 0.02 μg of T4 gene protein 32 (QBiogene, Illkirch, France), 1X of QuantiTect SybrGreen PCR Master Mix (Qiagen, Courtaboeuf, France) and 12.5 ng of soil DNA extract or 10^2 - 10^7 copies of standard DNA (*Pseudomonas stutzeri* ATCC 14405). The samples were run in duplicate as follows: 15 min at 95°C, 40 amplification cycles (15 s at 95°C, 30 s at 66°C, 30 s at 72°C and 15 s at 80°C) and 10 s at 40°C. Melting curves analysis confirmed the amplification specificity.

Diversity of bacteria, archaea and ammonia-oxidizers

High-throughput sequencing of bacterial and archaeal *16S rDNA* genes and *amoA* AOB and AOA genes (Table 1 and Table S1) were performed on an Illumina MiSeq[®] platform by Molecular Research DNA, USA. Sequencing was performed on the samples obtained after 7 or 90 days of incubation. A combination of the tools available from the RDP FunGene website (Fish *et al.*, 2013) and the open-source software Mothur (v.1.33.3) (Schloss *et al.*, 2009) was used to process and analyze the sequence data (Table 1). Sequencing products were first paired in overlapping pair-ends, except for *amoA* (AOA) amplicons that were too long (> 500 bp) for the MiSeq technology. Resulting sequence data were then sorted according to their length, and the quality of the primers (< 2 errors) and barcodes (< 1 error). The primers and barcodes were trimmed off before searching potential chimeric formation using UCHIME (Edgar *et al.*, 2011) implemented in Mothur. Putative chimeras were removed from the dataset. For *amoA* sequences, nucleotide sequences were translated in amino-acids and possible frame-reading shifts were detected and corrected using the FRAMEBOT algorithm (Wang *et al.*, 2013).

We clustered the corrected *16S* sequences into operational taxonomic units (OTUs) by setting a 0.03 distance limit (Kim *et al.*, 2011). The *amoA* sequences were clustered at 0.05 dissimilarity. The taxonomic identification of bacterial and archaeal *16S rDNA* genes was performed using the Greengenes 13.5 database. Similarity of *amoA* sequences with known references were assessed with 7 652 sequences of *amoA* AOB and 9 326 sequences of *amoA* AOA extracted from the FunGene database. The relatedness of *amoA* AOA sequences with identified archaeal taxa was performed using the database assembled by Pester *et al.* (2012). Rarefaction curves based on identified OTU and species richness estimator ACE were generated using Mothur for each sample. Singleton reads were not considered for subsequent analyses. Non-metric Multi-Dimensional Scaling (NMDS) were performed using the *metaMDS* function, available in the Vegan package (Oksanen *et al.*, 2007) of the R software (R Core Team, 2015) based on Bray-Curtis distances, associated to Permanova (Permutational multivariate analysis of variance) using the *adonis* function, to explore the difference in community composition in the different treatments.

Table 1 Main information on the diversity analysis of bacterial, archaeal, AOB and AOA communities

Community (Gene)	Primers	Mean Fragment size	Mean sequence number per sample	Total sequence number analyzed
Bacteria (16S rDNA)	515F / 806R (V4 Region)	300 bp	4580 ± 166	160301
Archaea (16S rDNA)	349F / 806R (V3-V4 Region)	424 bp	5745 ± 3332	206831
AOB (amoA)	amoA_1F / amoA_2R	435 bp	11745 ± 451	422844
AOA (amoA)	CrenamoA23F / CrenamoA616R	252 bp	16929 ± 9545	609456

Statistical analysis

All results are presented as means (\pm standard errors). A one-way analysis of variance (ANOVA) and *post-hoc* Tukey HSD were performed to test the effect of TiO₂-NP concentrations on measured variables at each sampling time. Where necessary, data were log-transformed prior to analysis to ensure conformity with the assumptions of normality and homogeneity of variances. Linear regressions were conducted to investigate relationships between all variables measured in the experiment, described by Spearman correlation coefficient (r). Effects with $P < 0.05$ are referred to as significant. All statistical analysis was carried out using the R statistical software 2.13.2 (R Core Team, 2011).

Path analysis (Shipley, 2002) was performed using Amos18[®] (Amos Development Corporation, Crawfordville, FL, USA) to explore the causal links between TiO₂ concentration, nitrifier abundance, denitrifier abundance, nitrification and denitrification activities. χ^2 test was used to evaluate model fit, by determining whether the covariance structures implied by the model adequately fit the actual covariance structures of the data. A non-significant chi-squared test ($P > 0.05$) indicates adequate model fits. The coefficients of each path as the calculated standardized coefficients were determined using the analysis of correlation matrices. These coefficients indicate by how many standard deviations the effect variable would change if the causal variable was changed by one standard deviation. The indirect effects of a variable can be calculated by the product of the coefficients along the path. Paths in this model were considered significant with a P -value < 0.05 .

RESULTS

Impact of TiO₂-NPs on soil functioning

NEA and DEA were considered as relevant proxies of soil functioning. Both TiO₂-NPs concentrations reduced NEA compared to the control after 30 days with 31% decrease ($P < 0.001$) and 20% ($P = 0.02$) for NPs concentrations of 1 and 500 mg.kg⁻¹, respectively (Figure 1A). These reductions were stronger after 90 days of incubation with about 40% decrease ($P < 0.001$) for both concentrations.

DEA was significantly decreased (Figure 1B) at the 3 sampling dates at 500 mg kg⁻¹ (7 days: - 15%, $P = 0.01$; 30 days: - 21%, $P = 0.04$; 90 days: - 19%, $P < 0.001$). At 1 mg kg⁻¹, a significant reduction of DEA was only observed after 90 days of incubation (- 15%, $P < 0.001$) in comparison with the control.

Impact of TiO₂-NPs on N-related functional guild abundances

Overall, the abundance of *amoA* AOA gene was 75 fold higher than the *amoA* AOB abundances in the studied soil, with an average of 9.27×10^7 and 1.25×10^6 copies for AOA and AOB, respectively. The dynamics of *amoA* AOA and *amoA* AOB under TiO₂-NP treatments were contrasted (Figure 2). While the application of 500 mg kg⁻¹ TiO₂-NPs lead to a decrease of *amoA* AOB gene abundance only after 90 days (Figure 2A, - 39 %, $P = 0.05$), no variation was observed at 1 mg kg⁻¹ during the course of the experiment. At the opposite, the abundance of *amoA* AOA was strongly decreased whatever the concentration after 90 days (Figure 2B, dose 1: - 53 %, $P < 0.001$; dose 500: - 60 %, $P < 0.001$). NEA was negatively correlated with the abundance of *amoA* AOB ($R^2 = 0.18$, $P = 0.002$, Table 2) but was positively correlated with the abundance of *amoA* AOA ($R^2 = 0.26$, $P < 0.001$, Table 2).

The abundance of *nirS* gene was 4.5 fold higher than the abundance of *nirK*, with an average of 1.15×10^7 and 2.57×10^6 copies per gram of dry soil, respectively. The *nirK* and *nirS* genes abundances were not significantly modified in the microcosms exposed to TiO₂-NPs over the course of the experiment (Figure 2C and 2D). DEA was not correlated to the abundance of *nirS* but was positively correlated with the abundance of *nirK* ($R^2 = 0.28$, $P < 0.001$, Table 2).

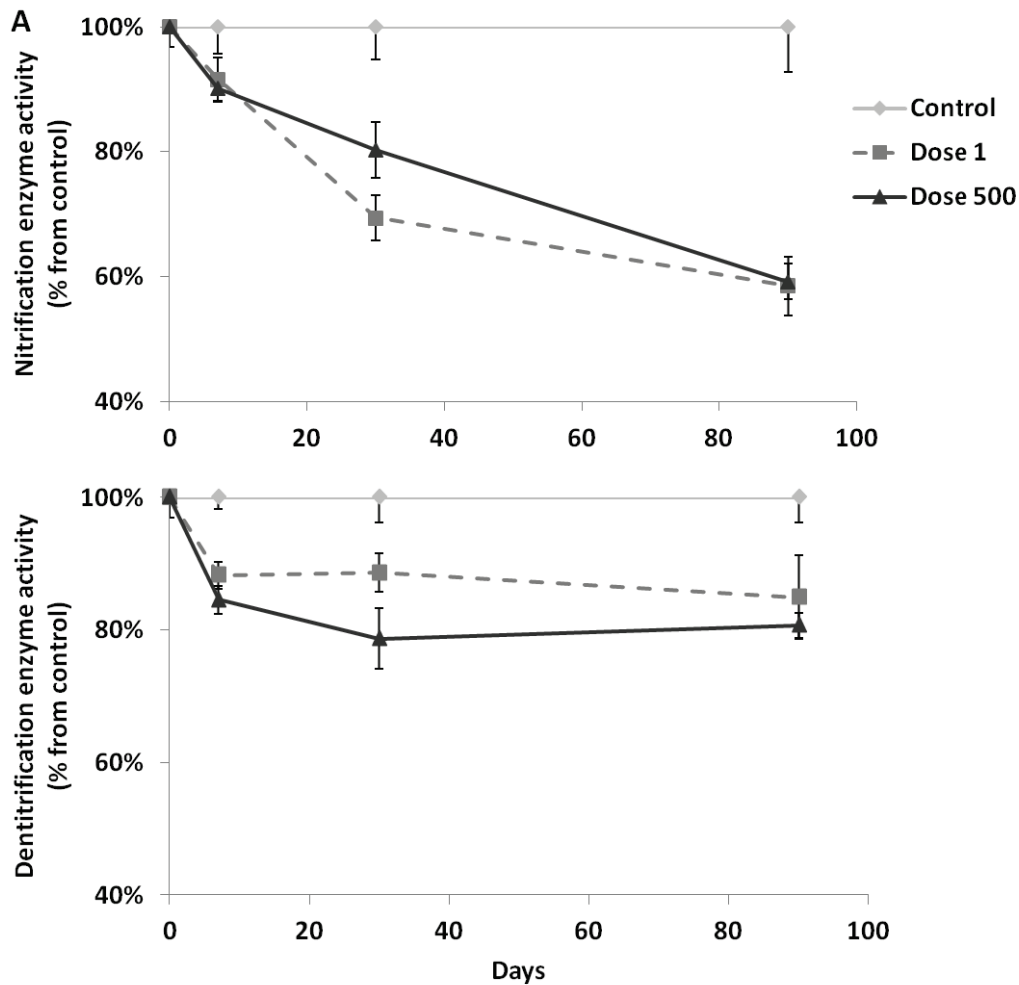


Figure 1 Changes over time of NEA (A) and DEA (B) in the different treatments (grey line: Control, dotted line: Dose 1 mg kg⁻¹, black line: Dose 500 mg kg⁻¹ dry soil). Error bars represent the standard error (n=6).

Table 2 Correlation table between the different measured variables (AOB abundance, AOA abundance, *nirK* abundance, *nirS* abundance, NEA and DEA). The strength of the linear relationship is given by the correlation coefficient *r* and negative numbers correspond to negative correlations, *P*-values are given in parenthesis. Significant correlations are presented in bold. Spearman correlations were investigated on all data of the 3 sampling times (n=54).

	AOB	AOA	<i>nirK</i>	<i>nirS</i>	NEA
AOB					
AOA	0.06 (0.69)				
<i>nirK</i>	0.04 (0.79)	-0.02 (0.94)			
<i>nirS</i>	0.22 (0.11)	0.27 (0.03)	-0.17 (0.20)		
NEA	-0.43 (0.002)	0.51 (<0.001)	0.35 (0.02)	0.01 (0.97)	
DEA	-0.39 (0.004)	0.20 (0.10)	0.53 (<0.001)	-0.08 (0.69)	0.68 (<0.001)

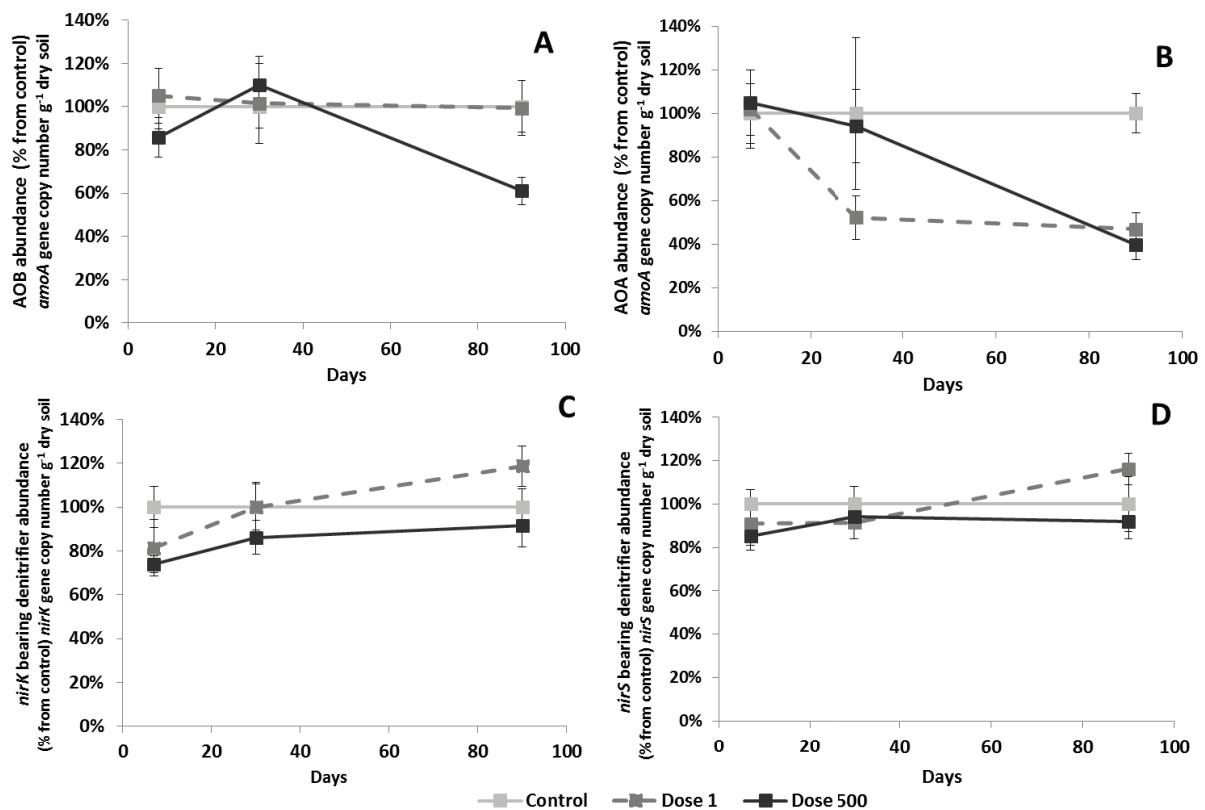


Figure 2 Changes over time of (A) ammonia-oxidizing bacteria (AOB), (B) ammonia-oxidizing archaea (AOA), (C) *nirK* bearing denitrifier abundance and (D) *nirS* bearing denitrifier abundance in the different treatments (grey line: Control, dotted line: Dose 1 mg kg⁻¹, black line: Dose 500 mg kg⁻¹ dry soil). Error bars represent the standard error (n=6).

Impact of TiO₂-NPs on soil microbial diversity

We assessed the impact of TiO₂-NPs on soil microbial diversity by sequencing the *16S rDNA* and *amoA* bacterial and archaeal genes.

160 301 original *16S rDNA* bacterial sequences yielded a total of 8 256 OTUs (average per sample: $1\,466 \pm 8.85$ OTUs) at a distance dissimilarity of 0.03. No significant change in richness (ACE estimator) was observed for this gene during the experiment and between treatments. However TiO₂-NPs contaminations, time and the interaction of time and exposure caused a significant shift in the bacterial community structure (TiO₂-NPs: $P < 0.001$, time: $P < 0.001$, TiO₂-NPs x time: $P = 0.005$, Figure 3A). Both doses of TiO₂-NPs significantly affected the bacterial community structure (Dose 1: $P = 0.001$, Dose 500: $P = 0.003$) after 90 days and the relative abundance of several OTUs were either lower (e.g. *Acidobacteria* group 6, *Saprospirae*) or higher (e.g. *Sulfuritalea*, *Anaerolinea*, *Bacteroidia*) than the control (Figure S1). After 90 days, the shift in the bacterial community structure induced by TiO₂-NP contamination was associated to a modification of the relative abundance of different phyla, especially the *Proteobacteria*, *Acidobacteria*, *Actinobacteria* and *Chloroflexi* phyla (Figure S2).

The archaeal community diversity assessed by *16S rDNA* gene sequencing exhibited a low richness with a total of 364 OTUs (average per sample: 28.75 ± 0.91 OTUs) from 206 831 original sequences at a dissimilarity distance of 0.03. In all samples, the community was dominated by a single OTU (relative abundance in average per sample: $70.4 \% \pm 2 \%$) belonging to an ammonia-oxidizing archeon of the genus *Nitrososphaera* (Figure S3). The archaeal community structure was modified during the incubation (time: $P = 0.002$, Figure 3C), but not in association with any TiO₂-NP exposure ($P = 0.34$). However, the number of sequences belonging to the dominant *Nitrososphaera* was significantly reduced in the microcosms exposed to both concentrations of TiO₂-NPs after 90 days (Dose 1: - 58 %, $P = 0.04$; Dose 500: - 59 %, $P = 0.04$, Figure 4A). Interestingly, the abundance of *amoA* AOA and the NEA were positively correlated with the number of *16S rDNA* gene sequence representing this dominant *Nitrososphaera* OTU (Figure 4C and 4D, respectively). After 90 days under TiO₂-NP exposure, while the number of *16S rDNA* gene sequence of this

dominant *Nitrososphaera* OTU decreased, the number of 16S rDNA gene sequences of the OTU belonging to the *Methanocella* genus was increased (relative abundance, in control: 0.3 % ± 0.08 %; in TiO₂-NP treatments 12 % ± 3 %, Figure S3).

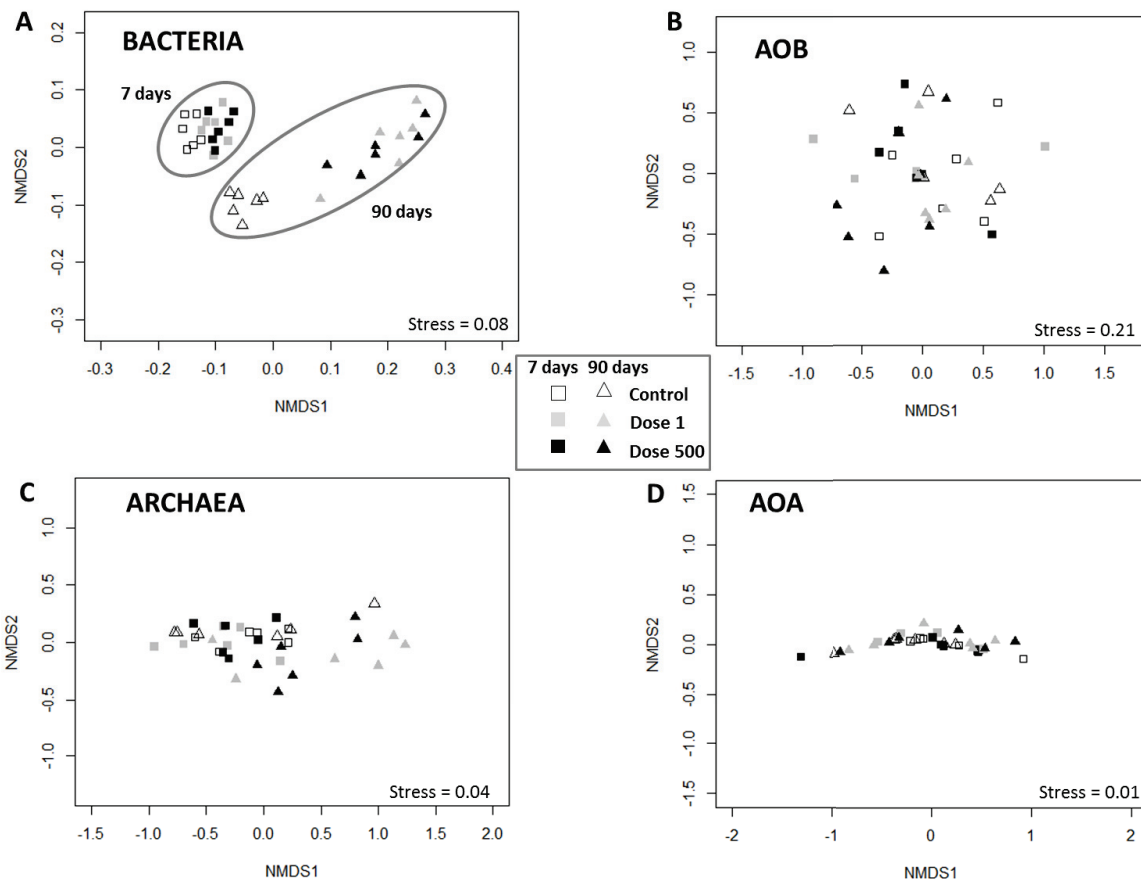


Figure 3 Nonmetric Multidimensional Scaling analysis to determine the modification in microbial community structure in presence of TiO₂-NPs (grey: Control, blue: Dose 1, red: Dose 500) after 7 days (square symbols) and 90 days (triangle symbols): A) Bacterial community, B) AOB community, C) Archaeal community, D) AOA community.

422 844 original *amoA* AOB gene sequences clustered into 5974 total clusters (average per sample: 272 ± 2.5 clusters) at a dissimilarity of 0.05. The ACE richness estimator of the AOB community was significantly decreased after 90 days in presence of 500 mg kg⁻¹ of TiO₂-NPs (- 8.4 %, *P* = 0.004). However the AOB community structure did not vary in the microcosms exposed to TiO₂-NPs throughout the incubation (TiO₂-NPs: *P* = 0.68, time: *P* = 0.21, Figure 3B).

609 456 original *amoA* AOA gene sequences yielded 2103 total clusters (average per sample: 44 ± 8 clusters) at a dissimilarity of 0.05. The AOA community structure assessed by *amoA* AOA gene sequences was not modified when exposed to TiO₂-NPs or during the incubation (TiO₂-NPs: $P = 0.95$, time: $P = 0.31$, Figure 3D). In all samples, similar to the results of 16S *rDNA* sequences, the *amoA* AOA sequences were dominated by a single cluster belonging to *Nitrososphaera* cluster (Figure S3), which represent in average 91 % of all sequences analyzed. Moreover, the other clusters obtained were also affiliated to this cluster (Figure S3). Contrary to the dominant 16S *rDNA Nitrososphaera* OTU, the number of sequences belonging to the dominant *amoA Nitrososphaera* cluster was not affected by TiO₂-NPs exposure at 7 or 90 days (Figure 4B). Furthermore, this dominant *amoA Nitrososphaera* cluster was not correlated to NEA, AOA abundance or to the dominant 16S *Nitrososphaera* OTU sequence number (data not shown).

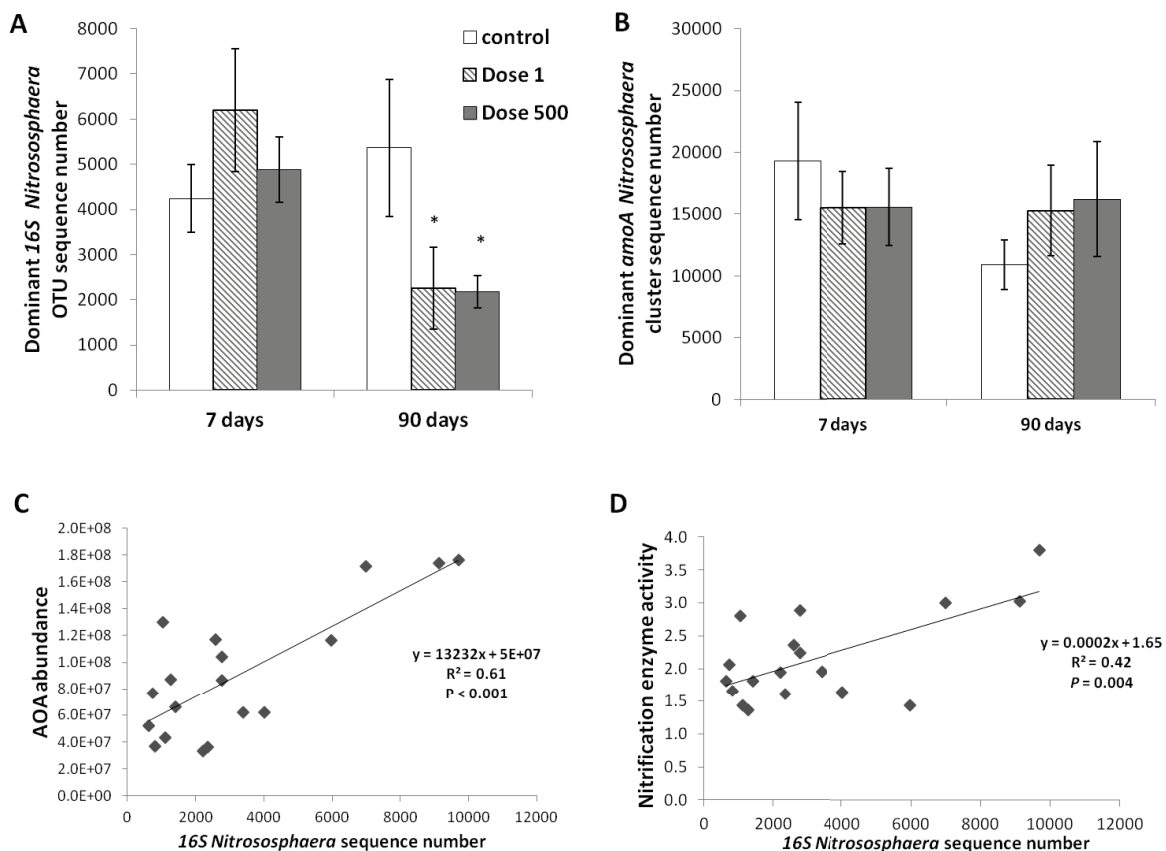


Figure 4 Effects of TiO₂-NPs A) on the dominant archaeal 16S *rDNA* OTU affiliated to the *Nitrososphaera* genus in the different treatments, B) on the dominant AOA *amoA* cluster

affiliated to the *Nitrososphaera* cluster (white bars: Control, hashed bars: Dose 1 mg kg⁻¹, black bars: Dose 500 mg kg⁻¹ dry soil). C) Correlation between the dominant 16S *Nitrososphaera* sequence number and AOA abundance (n=36). D) Correlation between the dominant 16S *Nitrososphaera* sequence number and NEA. The regression is drawn and the R² and P-values are given.

Potential causal relationships between TiO₂ contamination, microbial communities and soil functioning

We investigated potential causal relationships between TiO₂ contamination and microbial activity, abundance and diversity using path analysis. Path analysis was performed from the data obtained after 90 days of exposure to TiO₂-NPs because their effects were the greatest on microbial activities and abundances (Figure 1 and 2). The variables included in the model were significantly correlated to NEA or DEA. The model explained a significant part of NEA (70 %) and DEA (75 %) variances. TiO₂-NPs significantly and negatively affected the abundance of the dominant 16S *Nitrososphaera* OTU (path coefficient = - 0.28, *P* = 0.02) and also of *amoA* AOA and AOB genes (path coefficient = - 0.30, *P* = 0.03 and - 0.61, *P* = 0.001 respectively). The number of 16S *rDNA* gene sequences of the dominant archaeal *Nitrososphaera* was an important driver of the AOA abundance (path coefficient = 0.70, *P* < 0.001). The path analysis supported the relative abundance of *amoA* AOA genes as the major driver of NEA in this soil (path coefficient = 0.77, *P* < 0.001) and the relative abundance of *amoA* AOB as a minor actor of NEA (path coefficient = 0.17, *P* = 0.23). A great part of the variance of NEA was explained by indirect effects of the dominant 16S *Nitrososphaera* OTU (indirect effect = 0.53). TiO₂-NPs influenced DEA through direct effects (path coefficient = - 0.38, *P* = 0.007) and not indirect effects via *nirK* relative abundance (indirect effects = - 0.05). The *nirK* gene abundance was a driver of DEA (path coefficient = 0.28, *P* = 0.03) but DEA was mainly explained by NEA activity (path coefficient = 0.61, *P* < 0.001), suggesting that TiO₂-NP contamination primarily affects this process through a decrease of AOA abundance and NEA activity.

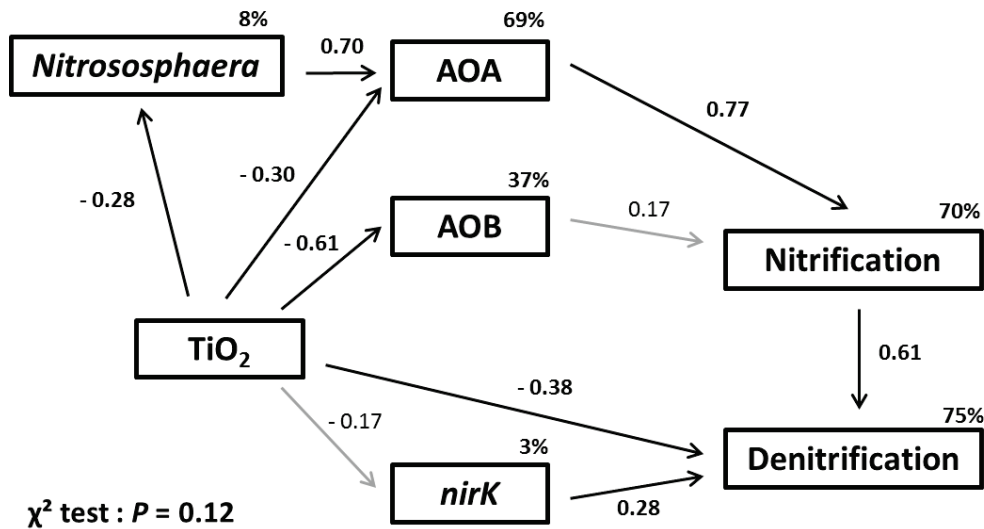


Figure 5 Path analysis of the direct and indirect effects of TiO₂-NPs, dominant *16S rDNA Nitrososphaera* sequence number, AOA abundance, AOB abundance, *nirK* abundance on NEA and DEA. Path coefficients (values indicated next to the arrows) correspond to the standardized coefficients calculated based on the analysis of correlation matrices and indicate by how many standard deviations the effect variable would change if the causal variable was changed by one standard deviation. Arrows and values in bold indicate a significant causal relationship between two variables.

DISCUSSION

Negative effect of a realistic concentration of TiO₂-NPs on the N cycle

TiO₂-NPs caused an important disturbance of the nitrogen cycle and a modification of the bacterial community structure in the agricultural soil studied even for a low realistic concentration (1 mg kg⁻¹ dry soil). Moreover, no resilience of the negative effects was observed during the time course of the experiment. Surprisingly, the lowest concentration had similar effects on potential activities and AOA abundance than the highest one (500 mg kg⁻¹ dry soil). The absence of linear dose-response of NPs on soil microbial activities has been previously observed (Ge *et al.*, 2011; Joško *et al.*, 2014; Simonin *et al.*, 2015). A hypothesis could be that the NPs homo- and heteroaggregation processes (i.e. the aggregation of NP with themselves and the aggregation of NP with other environmental constituents) are modified in function of the NPs concentration applied to soil (Lowry *et al.*, 2012) with

consequences on NP bioavailability and toxicity for microorganisms (Cornelis *et al.*, 2014). However, further work focusing on the physicochemical properties of NP in relation with the concentration applied into soil would be needed.

The absence of resilience of the denitrification activity and the increased over time declines of nitrification and ammonia-oxidizers abundance raise concerns about the ecotoxicity of TiO₂-NPs in soil. The greatest effects of TiO₂-NPs appeared 90 days after the contamination suggesting that aged NP can still be toxic to microorganisms even at low concentrations and after a long period of exposure. This should be considered in regards to transport experiments suggesting that TiO₂-NPs exhibit a low mobility in soils and would have a long residence time in these ecosystems (Fang *et al.*, 2009; Nickel *et al.*, 2015). Therefore, our results imply that short term experiments may have not accurately reflected the toxic potential of NP in soil. Most studies are based on short incubation periods not exceeding 60 days with high concentrations of NP (>100 mg kg⁻¹) (Simonin and Richaume, 2015). Considering these results, we strongly encourage further research to use realistic concentrations with longer incubations.

AOA are more affected by TiO₂-NPs than AOB

The AOA abundance was highly reduced by TiO₂-NPs contamination even at low concentration, especially compared to the AOB abundance. In this case, the AOA abundance decreases down to 60% after 90 days of incubation. Consistent with a previous study (Mertens *et al.* 2009), our results challenge the view that *Archaea* are more tolerant to chronic stresses than *Bacteria* (Schleper *et al.*, 2005; Valentine, 2007). The study of the ecology of AOA in soil and their response to environmental stressors is still at their infancy and very few studies have investigated the impact of pollutants on soil AOA (Schauss *et al.*, 2009; Mertens *et al.*, 2009; Liu *et al.*, 2010; Ollivier *et al.*, 2012, 2013) . To our knowledge, none studied the impacts of NPs. Pure culture studies have shown that TiO₂-NPs can be toxic to *Bacteria* after adsorption to cell membrane causing an oxidative stress associated to reactive oxygen species (ROS) production (Adams *et al.*, 2006; Simon-Deckers *et al.*, 2009). The very limited knowledge of soil *Archaea* physiology and the absence of toxicological

studies using soil archaeal strains do not allow hypothesizing that the same mechanisms are involved in AOA mortality in presence of TiO₂-NPs.

In the studied soil, AOA affiliated to the dominant *Nitrososphaera* genus were likely the main functional drivers of nitrification as suggested by the positive correlations with NEA and an AOA abundance 75-fold higher than AOB abundance. As illustrated by the path analysis, TiO₂-NPs had negative effects on AOA abundance with cascading negative effects on nitrification and denitrification activities. Thus, AOA *Nitrososphaera* had a pivotal role in the response of this soil to TiO₂-NP contamination.

A higher abundance of AOA than AOB in soil exhibiting neutral or alkaline pH have been reported several times (Leininger *et al.*, 2006; Jia and Conrad, 2009). Our results confirmed these observations and pointed out for the first time (to authors' knowledge), that AOA can dominate ammonia-oxidizing activity under alkaline pH and high OM content (Shen *et al.*, 2008; Jia and Conrad, 2009; Xia *et al.*, 2011; Ai *et al.*, 2013; Le Roux *et al.*, 2013).

In our study, the archaeal diversity was very low compared to the bacterial community and the most represented soil archaea (*16S rDNA*) and AOA (*amoA*) was affiliated to *Candidatus Nitrososphaera*. This observation is consistent with the literature reporting that this AOA is dominant in many agricultural soils worldwide (Taketani and Tsai, 2010; Bates *et al.*, 2011; Pester *et al.*, 2012; Zhalnina *et al.*, 2013). Interestingly, this dominant *16S rDNA Nitrososphaera* OTU decreased with TiO₂-NPs exposure and was positively correlated to NEA and AOA abundance. In contrast, the dominant *amoA Nitrososphaera* cluster was not affected by the contamination and was not related to AOA abundance, NEA or *16S rDNA Nitrososphaera* OTU. This result advocates for a lower level of diversity for *amoA* AOA genes than for *16S rDNA* AOA genes (Tourna *et al.*, 2008) and suggests that TiO₂-NPs exposure affected taxonomically different AOA groups harboring similar *amoA* genes. Accordingly for AOB community, we did not observed any particular pattern of *amoA* AOB community structure despite a broad shift in the bacterial community as assessed by *16S rDNA* gene diversity.

Cascading negative effects on denitrification

Unlike the nitrifier community, the abundance of the denitrifying community was not reduced by TiO₂-NPs exposure, yet DEA was constantly decreased (around – 20 %) all along the incubation. The higher resistance and resilience of denitrifiers to pollutants is well acknowledged and can be explained by their high functional redundancy, niche breadth and adaptive ability (Bissett *et al.*, 2013). The path analysis indicated that the decrease of denitrification activity was mainly explained by an indirect effect of the decline of nitrification caused by the decrease of the AOA community, which was related to TiO₂ exposure. However the decline of nitrification did not fully explain the denitrification decrease, because TiO₂-NPs also directly affected denitrification (path coefficient = - 0.38). This path reflects that other variables not included in the model are linked to the decrease of DEA. As suggested by Cantarel *et al.* (2012), specific clusters of denitrifying bacteria could be better related to denitrification than functional gene abundance (*nirK* and *nosZ*).

Conclusions

Nanomaterials such as TiO₂ are worrying emerging contaminants of terrestrial ecosystem. Their effects not only on soil functioning but also on microbial diversity and abundance of functional guilds are of particular interest for the environmental risk assessment. Our results suggest that direct effects do not explain NPs impact on N cycle. Considering functional links and indirect effects on key actors of this cycle provide an integrative assessment of their impact on the soil microbial community and its functioning. Particularly, soil nitrification and ammonia-oxidation performed by AOA, deserve deeper analyses from both a toxicological and environmental perspectives.

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Lyon), quantitative PCR at the DTAMB platform (IFR 41, University Lyon 1) and bioinformatic analyses at the Ibio platform (Microbial Ecology UMR5557-USC1364, Lyon).

CONFLICT OF INTEREST

The authors declare no conflict of interest.

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SUPPLEMENTARY INFORMATION

Table S1 Number of sequences processed during bioinformatic analyses and number of normalized sequences used for calculation of diversity indexes and correlations.

Community (Gene)	Total number of sequence before bioinformatic analysis	Number of sequence per sample normalized for analyses
Bacteria (<i>16S rRNA</i>)	3050615	4312
Archaea (<i>16S rRNA</i>)	1861424	1185
AOB (<i>amoA</i>)	5100153	9982
AOA (<i>amoA</i>)	1371316	2914

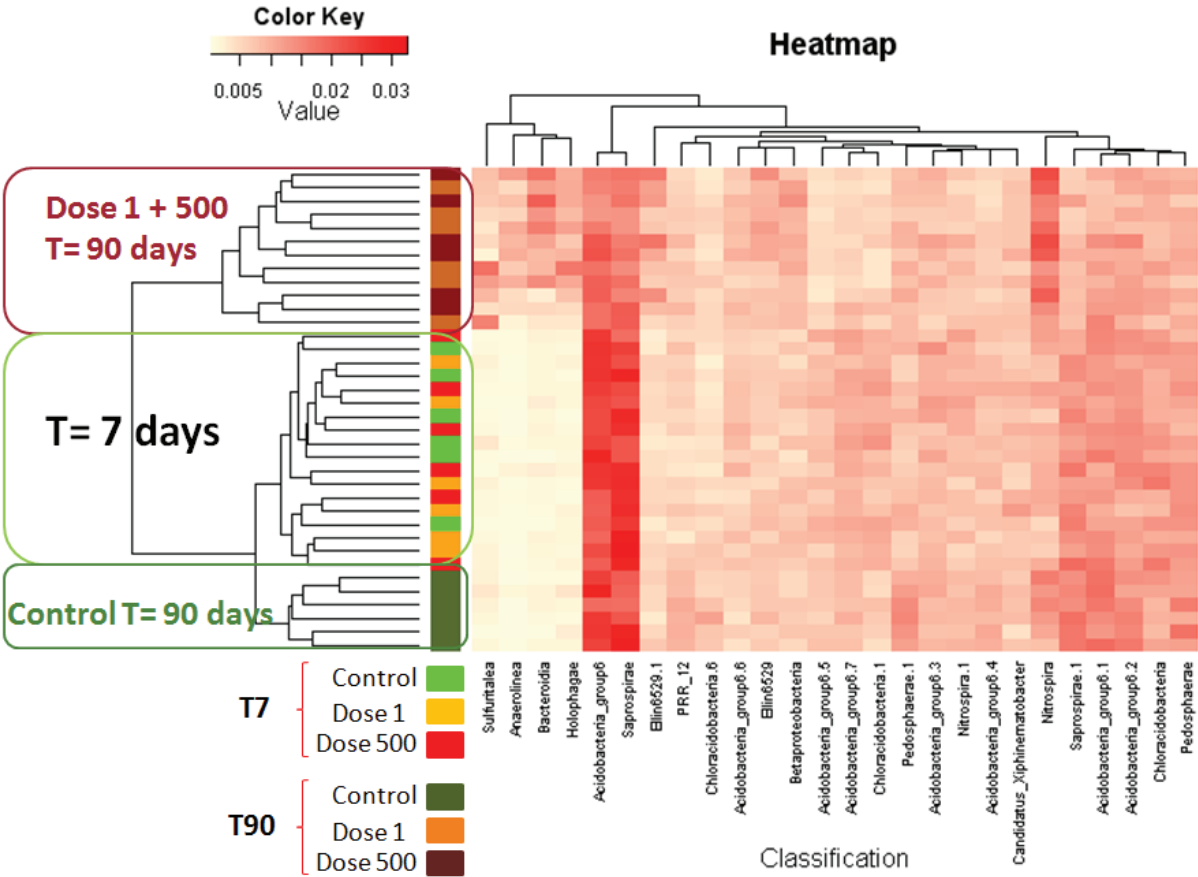


Figure S1 Relative abundance of bacterial OTUs in function of the different TiO₂-NPs concentrations and duration of incubation. The heatmap represents the proportions of OTUs at the genus level.

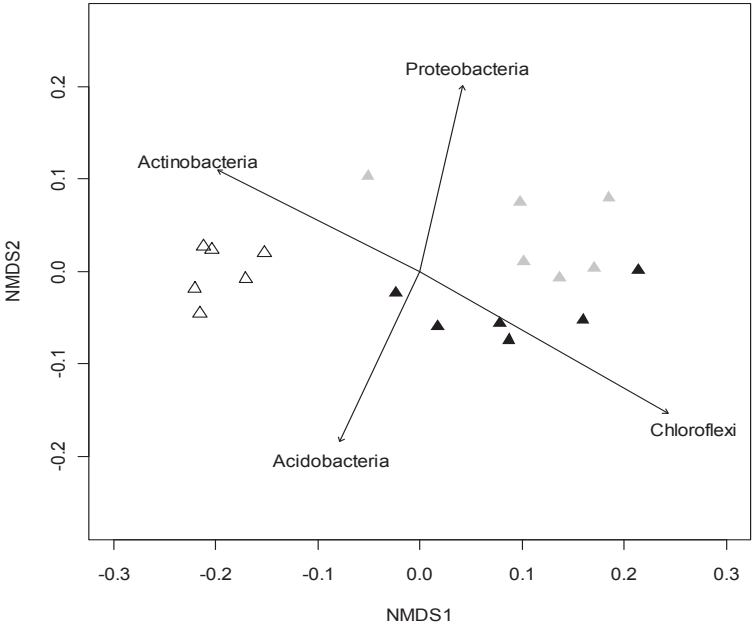


Figure S2 Nonmetric multidimensional scaling (NMDS) of the bacterial community structure after 90 days of exposure to TiO₂-NPs (control: white triangle; Dose 1: grey triangle; Dose 500: black triangle). Significant correlation between NMDS axis and bacterial phyla relative abundance, based on a post hoc permutation test (n = 999), are represented as arrows. The final stress for NMDS analysis was 0.06.

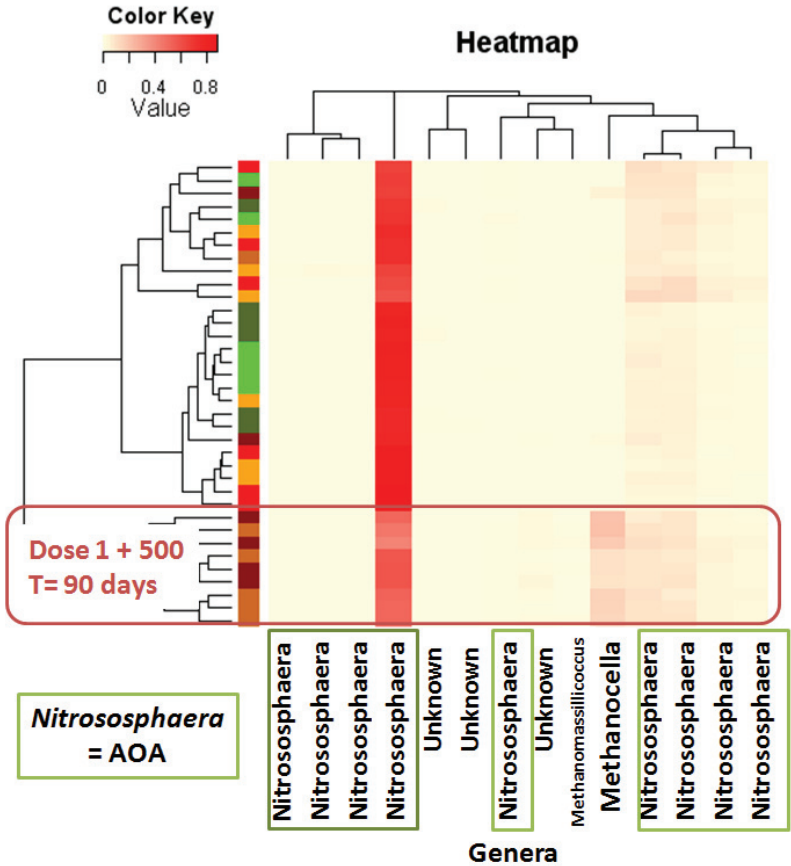


Figure S3 Relative abundance of archaeal genera in function of the different TiO₂-NPs concentrations and duration of incubation. The heatmap represents the proportions of OTUs at the genus level. See color legend on Figure S1.

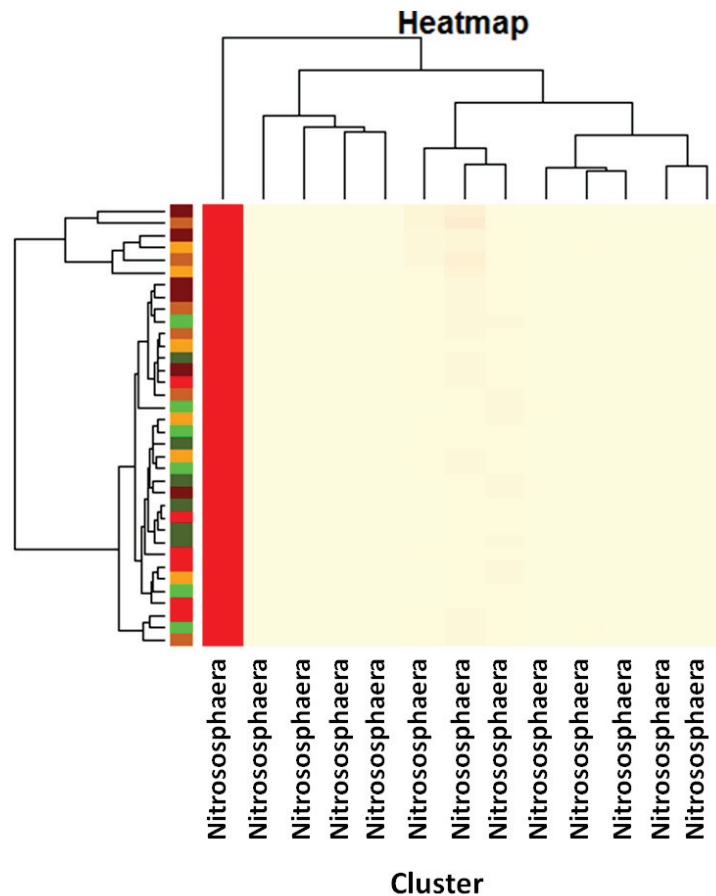


Figure S4 Relative abundance of AOA OTUs in function of the different TiO₂-NPs concentrations and duration of incubation. The heatmap represents the proportions of the different AOA clusters. See color legend on Figure S1.

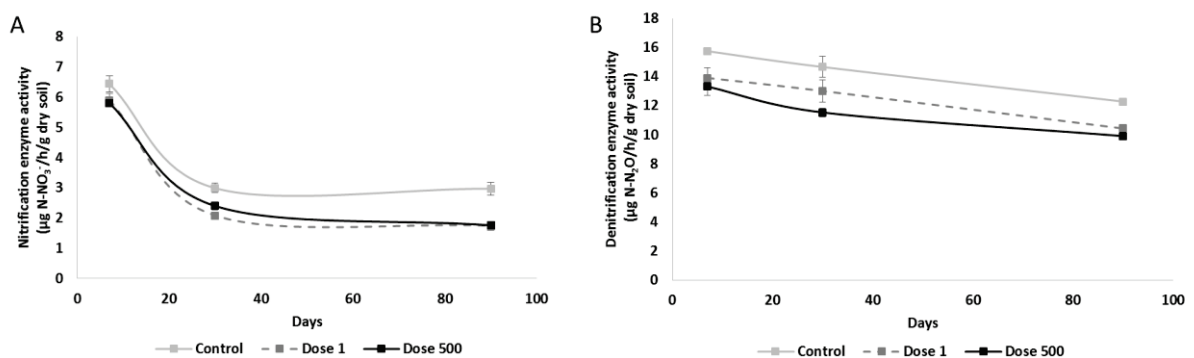


Figure S5 Dynamics over time of: A) NEA and B) DEA in the control (grey line) and in the Dose 1 mg kg⁻¹ (dotted line) and Dose 500 mg kg⁻¹ (black line) TiO₂-NPs treatments.

3. Etude de l'effet dose-réponse du TiO₂ sur l'activité nitrifiante et les groupes fonctionnels nitrifiants lors d'une contamination aigüe

a. Article 5 : Présentation générale de l'étude et synthèse des principaux résultats

Dans l'article précédent, nous avons mis en évidence que la nitrification est un processus microbien sensible aux TiO₂-NPs dans le sol limono-argileux étudié. Nous avons observé que tout comme pour la minéralisation du carbone, des effets négatifs similaires étaient observés pour des concentrations de 1 ou 500 mg kg⁻¹ sur les activités nitrifiantes et dénitrifiantes, ainsi que sur l'abondance des AOA. Cette absence de relation dose-réponse interpelle. C'est pourquoi nous avons approfondi cet aspect en évaluant l'influence d'une gamme de concentrations allant de 0,05 à 500 mg kg⁻¹ sur la nitrification en utilisant le sol limono-argileux comme sol modèle. Cette expérience a permis de déterminer s'il existait une relation dose-réponse dans le cas d'une contamination aux TiO₂-NPs dans un sol, comme cela est observé pour des polluants solubles et utilisé pour déterminer par exemple des concentrations efficaces médianes (EC50) dans le cadre de réglementations. De plus, nous avons étudié l'effet des TiO₂-NPs à des concentrations très faibles, rarement évaluées dans la littérature (Article 1), qui peuvent représenter des doses réalistes dans les sols.

Les effets du TiO₂ aux différentes concentrations ont été évalués sur l'activité nitrifiante, ainsi que sur l'abondance des 4 groupes fonctionnels microbiens impliqués (AOA, AOB, NOB *Nitrobacter* et *Nitrospira*). Dans l'article précédent, seuls les effets sur les AOA et AOB impliqués dans la 1^{ère} étape de la nitrification avaient été considérés et 30 % de la variation de la nitrification n'avait pas été expliquée dans la path analysis. La seconde étape réalisée par les NOB *Nitrobacter* et *Nitrospira* n'est pas considérée comme limitante pour la nitrification (Kowalchuk and Schauss, 2001). Toutefois une accumulation du nitrite a été observée à plusieurs reprises dans des sols perturbés, suggérant que la seconde étape réalisée par les NOB peut parfois également être limitante (Roux-Michollet *et al.*, 2001 ; Ollivier *et al.*, 2013). C'est pourquoi dans cette étude, nous nous sommes également intéressés à la réponse des NOB aux TiO₂-NPs pour comparer les effets observés sur l'abondance des 4 groupes de nitrifiants et comprendre les conséquences sur l'activité nitrifiante globale. Grâce à une path analysis, nous avons déterminé l'influence de cette

Chapitre 3 : Impact du TiO₂ sur le cycle de l'azote : exemple d'un sol limoneux-argileux

perturbation sur le couplage entre les microorganismes oxydant l'ammonium (AOA ou AOB) et ceux oxydant le nitrite (NOB *Nitrobacter* et *Nitrospira*), afin d'expliquer les modifications d'activité nitrifiante en fonction des concentrations testées.

Afin de mieux comprendre la toxicité observée pour chaque concentration, l'agrégation et le potentiel oxydant (production de ROS) des TiO₂-NPs ont été mesurés. Ces paramètres sont connus pour être impliqués dans la biodisponibilité et la toxicité de ces composés (cf. chapitre 1).

Après 90 jours d'exposition aux différentes doses de TiO₂-NPs, nos résultats montrent des effets plus importants sur le groupe des AOA, alors que *Nitrospira* semble insensible à cette perturbation. L'abondance des AOB et des *Nitrobacter* a été modifiée de façon similaire et l'abondance de ces 2 groupes était fortement corrélée positivement. La nitrification potentielle a été réduite significativement à la concentration la plus faible (0.05 mg kg⁻¹) et aux concentrations les plus fortes (100 et 500 mg kg⁻¹).

Quelle que soit la variable mesurée, aucune relation dose-réponse linéaire n'a été observée et des pourcentages de diminution similaires ont été constatés pour des concentrations différant de plusieurs ordres de grandeur. Cette observation a pu être expliquée en partie par le fait que les caractéristiques physico-chimiques des TiO₂-NPs sont modifiées en fonction de la concentration utilisée. En particulier, nous avons pu mettre en évidence que l'agrégation du TiO₂ augmentait avec la concentration, alors que le potentiel oxydant diminuait. Ces résultats expliqueraient les diminutions significatives de l'activité nitrifiante ainsi que celles de l'abondance des AOA et *Nitrobacter* pour les concentrations testées les plus faibles (0.05 et 0.1 mg kg⁻¹). A ces faibles concentrations, les NPs sont moins agrégées et ont un potentiel oxydant plus grand ce qui très certainement leur confère une toxicité plus importante.

Via une approche par path analysis, nous avons montré que les altérations de la nitrification pouvaient être expliquées par des modifications de couplage entre les groupes fonctionnels impliqués, associées à une sensibilité différente de ces 4 groupes à cette perturbation. Ces résultats mettent en évidence une redondance fonctionnelle surprenante au sein des microorganismes oxydant l'ammonium et le nitrite, qui a permis de maintenir l'activité nitrifiante à des niveaux similaires à celui du contrôle pour la majorité des doses testées.

En conclusion, ces travaux mettent en évidence que les TiO₂-NPs peuvent avoir des effets délétères sur des processus microbiens clés à des concentrations très faibles dans les sols. Ils soulignent également la faible pertinence de l'utilisation des relations dose-réponses classiquement utilisées en écotoxicologie pour évaluer la toxicité des NPs dans les sols.

b. « Soil nitrification is altered by titanium dioxide nanoparticles through modifications of coupling between ammonia- and nitrite-oxidizers »

L'article 5 intitulé «Soil nitrification is altered by titanium dioxide nanoparticles through modifications of coupling between ammonia- and nitrite-oxidizers» a été soumis dans le journal *FEMS Microbiology Ecology* le 23 juin 2015.

Soil nitrification is altered by titanium dioxide nanoparticles through modifications of coupling between ammonia- and nitrite-oxidizers

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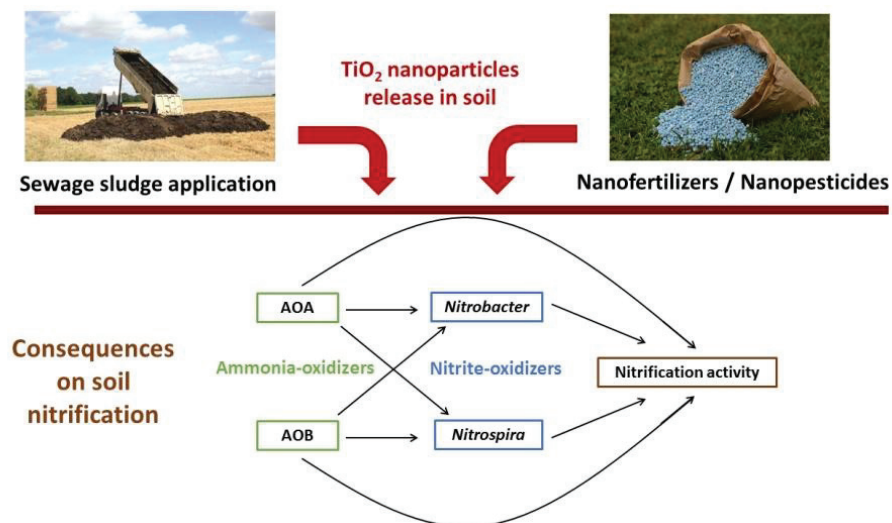
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RUNNING TITLE Soil nitrification sensitivity to TiO₂ nanoparticles

GRAPHICAL ABSTRACT



ABSTRACT

Nitrification is a crucial microbiological process for soil fertility depending on the combined activity of ammonia- and nitrite-oxidizers. This key step of the N cycle is known to be very sensitive to environmental stressors. However, the impact of emerging pollutants such as titanium-dioxide nanoparticles (TiO₂-NPs) which are increasingly released in agricultural soils, remains to be investigated.

We assessed the impact of eight TiO₂-NPs concentrations (0.05-500 mg kg⁻¹ dry-soil) in soil microcosms on potential nitrification activity and abundance of ammonia-oxidizing archaea (AOA), bacteria (AOB) and nitrite-oxidizers (*Nitrobacter* and *Nitrospira*) using quantitative PCR. In addition, aggregation and oxidative potential of TiO₂-NPs were measured to identify the main drivers of TiO₂-NPs toxicity in soil.

After 90 days of exposure, AOA appeared to be the most affected by TiO₂-NPs, while *Nitrospira* seemed insensitive to this contamination. AOB and *Nitrobacter* abundance exhibited similar responses. Interestingly, the potential nitrification activity was reduced by the lowest (0.05 mg kg⁻¹) and the highest (100 and 500 mg kg⁻¹) TiO₂-NPs concentrations. A path analysis suggested that the alteration of nitrification could be explained by changes in the coupling between ammonia- and nitrite-oxidizers triggered by a different sensitivity of these microbial groups to this contaminant.

Key-words: TiO₂ Nanomaterials, Microbial Ecotoxicology, Nitrification, Structural Equation Modeling

INTRODUCTION

Soils are constantly submitted to new environmental stressors, such as metal-oxide nanoparticles (NPs), which are worrying emerging pollutants. Metal-oxide NPs are present in many commercial products (food, cosmetics, paints, electronics...) and can be released in the environment all along their life cycle from production to disposal (Mitrano et al., 2015). Soils are particularly exposed to NPs that can be found in sewage sludge used as agricultural amendments or in nanofertilizers and nanopesticides (Brar et al., 2010; Servin et al., 2015). Titanium-dioxide (TiO₂) NPs are the most produced NPs and it is predicted that they would represent 50% of all NPs retrieved in soil (Keller et al., 2013; Sun et al., 2014). The few available studies have shown that TiO₂-NPs could have detrimental effects on soil microbial communities, as deduced from the decrease of soil respiration or bacterial diversity and modifications of soil bacterial community structure (Ge et al., 2011, 2012; Simonin et al., 2015a). In a recent review, we identified several research needs on NPs ecotoxicity (Simonin and Richaume, 2015). Among these, the assessment of their impact using a range of relevant environmental TiO₂-NPs concentrations is of interest since traditional toxicological assays to find out dose–response relationships are not workable in the case of nanoparticles (Editorial Nature Nanotechnol 2011). Given bacteria are recognized as important targets to consider when assessing NPs impact in soils, the consequence of NPs toxicity on microbial functional groups involved in key soil services such as fertility, is relevant in risk assessment studies. In this context, the microbial functional groups involved in the N cycle, especially nitrification, appear to be an ideal model (Wessén and Hallin, 2011; Pereira e Silva et al., 2012). Indeed, one can assume that the effect of NPs on microbial functional groups exhibiting low diversity and low functional redundancy such as nitrifiers could be detrimental to soil functioning.

Nitrification is crucial for plant nutrition and biomass production by controlling soil inorganic-N availability. This is also one of the most sensitive microbial process to disturbance, such as pollutant contamination (Dalzell et al., 2002; Broos et al., 2005). Nitrification is a two-step process performed by phylogenetically constrained groups of microorganisms. Ammonia-oxidation (oxidation of NH₄⁺ in NO₂⁻) is carried out by ammonia-oxidizing archaea (AOA) (Leininger et al., 2006) and ammonia-oxidizing bacteria (AOB)

(Kowalchuk and Stephen, 2001). Most of the literature about nitrification is focused on this step, as it is considered as the limiting-step of the whole nitrification (Kowalchuk and Stephen, 2001). Although AOA were discovered recently (Könneke et al., 2005), their functional role in nitrification is now well established (Prosser and Nicol, 2012; Zhahnina et al., 2012). Nevertheless, little knowledge is available about their sensitivity to environmental stressors (global change, pollutants...) (Mertens et al., 2009; Ruyters et al. 2010; Ollivier et al., 2012).

The second step of nitrification is nitrite-oxidation (oxidation of NO_2^- in NO_3^-) which is performed by nitrite-oxidizing bacteria (NOB), especially the *Nitrobacter* and *Nitrospira* genera in soil (Freitag et al., 2005). Compared to ammonia-oxidation, nitrite-oxidation is less studied, although efficient nitrification activity requires the presence of both ammonia-oxidizers and nitrite-oxidizers (Ollivier et al., 2013). In disturbed soils, nitrite can accumulate indicating that nitrite-oxidation could be the limiting-step of nitrification in some cases (Gelfand and Yakir, 2008; Roux-Michollet et al., 2008). However, nowadays the responses of AOA and NOB to stressors and the alteration of the coupling between ammonia-oxidizers and nitrite-oxidizers in disturbed soil systems is insufficiently documented (Ollivier et al., 2013).

In vitro NPs toxicity is attributed to an oxidative stress associated to the production of reactive oxygen species (ROS) from NPs in contact with microbial membranes, leading to membranes disruption, proteins oxidation, or energy transduction interruption (Klaine et al., 2008; Neal, 2008; Xia et al., 2008). However, these drivers of NPs toxicity have not been demonstrated in soil yet. This can be explain by (i) current technical limitations to detect NPs in complex media (Tourinho et al., 2012; Cornelis et al., 2014), (ii) the multiple and complex physicochemical transformations that NPs can undergo in soil, such as homo- and heteroaggregation, which are greatly influenced by environmental abiotic factors (e.g. pH, ionic strength, organic matter) (French et al., 2009; Keller et al., 2010; Lowry et al., 2012), (iii) the influence of NPs concentration on their physicochemical properties, especially on their aggregation and reactivity (Menard et al., 2011; Vitorge et al., 2013; Pokhrel et al., 2014). Thus, to date it is impossible to predict the impact of different NPs concentrations on soil organisms and little is known on the toxicological mechanisms involved.

In this study, we assessed the impact of a range of 8 TiO₂-NPs concentrations varying from 0.05 to 500 mg kg⁻¹ dry soil on the potential nitrification activity and the abundance of ammonia-oxidizers (AOA and AOB) and nitrite-oxidizers (NOB *Nitrobacter* and *Nitrospira*) using quantitative PCR. Concentrations ranging between 0.05 and 0.5 mg kg⁻¹ were considered as low concentrations; those ranging between 1 and 50 mg kg⁻¹ as intermediate concentrations and 100 and 500 mg kg⁻¹ as high concentrations. The experiment was performed with a silty-clay soil that was artificially contaminated in the laboratory and exposed to TiO₂-NPs for 90 days in microcosms. We assume that the responses to TiO₂-NPs of AOA, AOB, *Nitrobacter* and *Nitrospira* could differ because of their different physiology and ecology. A path analysis was used to determine if the effects on potential nitrification could be explained by the modification of nitrifier abundance and if the functional drivers of nitrification and coupling between ammonia- and nitrite-oxidizers were modified according to the TiO₂-NPs concentrations. In addition, we measured the aggregation and oxidative potential (total ROS production) of TiO₂-NPs in suspension in water versus soil solution at different concentrations. These data have been collected in order to identify and understand better the main drivers of TiO₂-NPs toxicity in soil and the effects on nitrification activity and nitrifier abundance.

MATERIALS AND METHODS

TiO₂ Nanoparticles

Titanium dioxide nanoparticles (TiO₂-NPs) were provided by Sigma Aldrich (St Louis, USA) (mixture of anatase (80%) and rutile (20%) crystal structure) with at least 99.5% purity. According to the manufacturer information, TiO₂-NPs presented a specific surface area of 35-65 m².g⁻¹ and a mean particle size of 21 nm in powder as measured by Transmission Electron Microscopy.

Soil microcosms and experimental design

Soil was collected from the upper 20 cm layer of a silty-clay soil (Cambisol, WRB, 2006) sampled at Commarin (Côte d'Or, France) under a permanent pasture. After collection, roots

and plant litter were manually removed. The soil was sieved (2 mm) and homogenized before storage at 4°C. The soil main characteristics were: sand 10%; loam 51%; clay 39%; pH 7.7; OM 7.9%; CEC 20.1 cmol kg⁻¹; Water holding capacity (WHC) 51%.

This soil was exposed to eight increasing TiO₂-NPs concentrations in microcosms for 90 days in the dark at 28°C. One kg of soil (equivalent dry weight – eq dw) was spiked with a defined volume of TiO₂-NPs suspensions prepared in ultrapure water (0.625 to 6250 mg TiO₂ L⁻¹) in order to obtain 0.05 to 500 mg TiO₂ kg⁻¹ dry soil (Table 1) and to increase the soil water content to the WHC. These concentrations were chosen to represent different exposures from a very low realistic environmental concentration (0.05 mg kg⁻¹ dry soil) to a high concentration representing an accidental spill (500 mg kg⁻¹ dry soil) (Sun et al., 2014). The TiO₂-NPs suspensions were added homogeneously to soil using a multichannel pipette and then soils were thoroughly mixed for 10 minutes to ensure a uniform spiking. Soils receiving only the same volume of ultrapure water were used as controls. Microcosms were set up by placing 50 g of control or spiked soils (eq dw) into 150 mL glass plasma flasks sealed with rubber stoppers to avoid soil drying and to maintain the soil moisture constant during the experiment. Microcosms were weekly aerated for 5 minutes to ensure a renewal of the atmosphere in the flasks. This experimental design resulted in a total of 36 microcosms with 4 replicates per treatment. At the end of each incubation time, microcosms were subsampled as follows: 3 g of soil (eq dw) were immediately used for the measurements of potential nitrification, 3 g of soil were stored at -20°C before DNA extraction.

Table 1 Correspondences between TiO₂ concentration in spiking suspensions and the final TiO₂ concentration in soil microcosms

Concentration TiO ₂ :	in spiking suspension (mg L ⁻¹)	0.625	1.25	6.25	12.5	125	625	1250	6250
	in soil (mg kg ⁻¹ dry soil)	0.05	0.1	0.5	1	10	50	100	500

Soil solution preparation

Soil solutions were prepared following the protocol described by Simonin et al. (2015a) to characterize TiO₂-NPs in physicochemical conditions as close as possible to those

encountered in soil in terms of pH, ionic strength and dissolved components. Soil colloids larger than the initial size of TiO₂-NPs (i.e. 20 nm) were removed from the suspension in order to get a reliable measurement of NPs aggregation and oxidative potential. Briefly, soil solutions were prepared by shaking 10 g of uncontaminated soil dispersed in 50 mL of ultrapure water during 30 min (180 rpm, 20°C) in a refrigerated incubator shaker (New Brunswick – Eppendorf, Hamburg, Germany). The solutions were then centrifuged for 20 min (8000 g, 20°C, Centrifuge 5804R, Eppendorf, Hamburg, Germany) and the supernatants were collected and stored at 4°C.

Aggregation and oxidative potential of TiO₂-NPs

TiO₂-NPs suspensions were prepared in ultrapure water or in soil solutions at different concentrations ranging from 0.625 to 125 mg L⁻¹. Higher concentrations of TiO₂-NPs were tested but did not allow to obtain reliable measurements of aggregation or oxidative activity, because of sample fluorescence and coloration. All TiO₂-NPs suspensions were dispersed through ultrasonication for 5 min before measurements to ensure suspension homogeneity. The apparent hydrodynamic diameter of TiO₂-NPs in ultrapure water or soil solutions was determined by dynamic light scattering (DLS) using a NanoZS Zetasizer (Malvern Instruments –UK) as described in Vitorge et al. (2014). For each suspension, the mean of 3 measurements was recorded.

The oxidative potential of TiO₂-NPs in ultrapure water and soil solution was assessed using the 2',7'-dichlorofluorescein (DCFH) assay, following Foucaud et al (2007). DCFH is a non-fluorescent reagent that becomes fluorescent dichlorofluorescein (DCF) when oxidized in the presence of ROS.

DCFH was prepared from 2',7'-dichlorofluorescein diacetate (DCFH-DA) according to the method provided by Cathcart et al. (1983). We mixed 0.5 mL of 1 mM DCFH-DA in methanol with 2 mL of 0.01M sodium hydroxide. The hydrolysate was allowed to stand at room temperature for 30 min and then neutralized with 10 mL phosphate buffer (KH₂PO₄, 0.1M). The solution was kept on ice avoiding any light exposure until use.

The measurement of DCFH oxidation by TiO₂-NPs was performed in 48-wells plates in which were added: 200 µL of DCFH and 600 µL of the different TiO₂-NPs suspensions prepared in

ultrapure water or in soil solution. Ultrapure water or soil solution was used as negative control and a 0.5M H₂O₂ solution was used as positive control. All assays were performed in triplicate. The fluorescence generated by the DCFH oxidation by ROS was measured in a microplate reader (Tecan Infinite M200) at 485 nm excitation and 530 nm emission, every 10 minutes during one hour.

DNA extraction and quantification of the abundance of nitrifiers

DNA was extracted from 0.5 g of frozen soil using the Power Soil™ DNA Isolation Kit (MO BIO laboratories, Carlsbad, CA, USA), following the manufacturer's instructions and then DNA concentrations were determined using the Qubit dsDNA BR Assay (Invitrogen).

The abundance of the ammonia-oxidizers (AOA and AOB) and nitrite-oxidizers (*Nitrobacter* and *Nitrospira*) were measured by quantitative PCR using a Lightcycler 480 (Roche Diagnostics, Meylan, France). For AOA and AOB quantification, the final reaction volume was 20 µL and contained (final concentrations) 0.5 µM of each primer for the bacterial *amoA* or 0.75µM of CrenamoA616r and 1µM of CrenamoA23f for the archaeal *amoA*, 2% bovine serum albumin (BSA), 1X of QuantiTect SybrGreen PCR Master Mix (Qiagen, Courtaboeuf, France) and 10 ng of soil DNA extract or 10⁷-10² gene copies number µL⁻¹ of an in-house plasmid containing cloned bacterial (*Nitrosomonas europaea*, GenBank accession number:L08050) and archaeal (54d9 fosmide fragment, Treusch et al., 2005) *amoA* genes.

For *Nitrobacter*-like NOB quantification, the final reaction volume was 20 µL and contained 0.5 µM of each primer, 1X of QuantiTect SybrGreen PCR Master Mix and 40 ng of soil DNA extract or 10⁷-10² copies using a linearized plasmid containing cloned *nxrA* gene of *Nitrobacter hamburgensis* X14 (DSMZ 10229).

For *Nitrospira* quantification the final reaction volume was 25 µL and contained (final concentrations) 0.4 µM of each primer, 1X of QuantiTect SybrGreen PCR Master Mix and 10 ng of soil DNA extract or 10⁷-10² *Nitrospira* copies of linearized plasmid DNA (GenBank accession number: FJ529918). All primers and thermal cycling conditions used are described in Table 2.

Melting curve analysis confirmed the specificity of amplification of the four genes. High amplification efficiencies of 96–98% were obtained.

Table 2 PCR primers and thermal cycling conditions used for quantification of ammonia- and nitrite-oxidizer abundance

Primers	Target gene	Reference	Thermal conditions
CrenamoA23f	AOA <i>amoA</i>	Tourna et al. 2008	15 min at 95°C, 45 cycles (45 s at 94°C, 45 s at 55°C and 45 s at 72°C)
CrenamoA616r	AOA <i>amoA</i>	Tourna et al. 2008	
amoA-1F	AOB <i>amoA</i>	Rotthauwe et al. 1997	15 min at 95°C, 45 cycles (30 s at 95°C, 45 s at 54°C, 45 s at 72°C and 15 s at 80°C)
amoA-2R	AOB <i>amoA</i>	Rotthauwe et al. 1997	
F1norA	NOB <i>Nitrobacter nxrA</i>	Poly et al. 2008	15 min at 95°C, 45 cycles (30 s at 95°C, 45 s at 55°C and 45 s at 72°C)
R2norA	NOB <i>Nitrobacter nxrA</i>	Wertz et al. 2008	
Ns675f	NOB <i>Nitrospira 16S</i> <i>rRNA</i>	Graham et al. 2007	15 min at 95°C, 45 cycles (30 s at 95°C, 30 s at 66°C and 1 min at 72°C)
Ns746r	NOB <i>Nitrospira 16S</i> <i>rRNA</i>	Graham et al. 2007	

Nitrification activity

Potential nitrification activity was determined following the protocol described in Dassonville et al. (2011). Sub-samples of fresh soil (3 g equivalent dry soil) were incubated with 6 ml of a solution of (NH₄)₂SO₄ (50 µg N-NH₄⁺ g⁻¹ dry soil). Distilled water was added in each sample to reach 24 ml of total liquid volume in flasks. The flasks were sealed with Parafilm® and incubated at 28°C with constant agitation (180 rpm). During incubation, 1.5 ml of soil slurry was sampled at 2h, 4h, 6h, 8h and 10h, filtered (0.2 µm pore size) and transferred into vials. Samples were stored at -20°C until analysis of NO₃⁻ concentrations by ionic chromatography (DX120, Dionex, Salt Lake City, USA) equipped with a 4 mm×250 mm column (IonPac AS9 HC). Potential nitrification was expressed as µg N-NO₃⁻ h⁻¹ g⁻¹ dry soil.

Statistical analysis

All results are presented as means (± standard error). A one-way analysis of variance (ANOVA) and *post-hoc* Tukey HSD were performed to test the effect of TiO₂-NPs concentrations on measured variables. When necessary, data were log-transformed prior to analysis to ensure conformity with the assumptions of normality and homogeneity of variances. Linear regressions between all variables measured were investigated. Effects with

$P < 0.05$ are referred to as significant. These statistical analyses were carried out using R statistical software 2.13.2 (R Core Team, 2015).

Path analysis was performed using Amos18[®] (Amos Development Corporation, Crawfordville, FL, USA) to explore the causal links between nitrifier abundance (AOA, AOB, *Nitrobacter* and *Nitrospira*) and nitrification rates (Shipley, 2002). This path analysis enabled to determine which groups of ammonia-oxidizers and nitrite-oxidizers are the main functional drivers of nitrification in our soil and if the drivers of nitrification are modified in function of the TiO₂-NPs concentration applied.

Four path analysis were performed using different sets of data: i) all concentrations of TiO₂-NPs, ii) only low concentrations of TiO₂-NPs (0.05; 0.1 and 0.5 mg kg⁻¹), iii) only intermediate concentrations (1; 10; 50 mg kg⁻¹) and iv) only high concentrations of TiO₂-NPs (100 and 500 mg kg⁻¹). A χ^2 test was used to evaluate model fit by determining whether the covariance structures implied by the model adequately fit the actual covariance structures of the data. The coefficients of each path as the calculated standardized coefficients were determined using the analysis of correlation matrices. These path coefficients indicate how many standard deviations the effect variable would change, if the causal variable was changed by one standard deviation. Paths in this model were considered significant with a $P < 0.05$.

RESULTS

Aggregation and oxidative potential of TiO₂-NPs

Aggregation and oxidative potential (OP) of TiO₂-NPs were characterized in the aqueous suspensions used to spike the soil (i.e. in ultrapure water) and in soil solutions prepared at TiO₂-NPs concentrations ranging from 0.625 to 125 mg L⁻¹.

In both ultrapure water and soil solutions, TiO₂-NPs aggregation measured by Dynamic Light Scattering (DLS) increased with NPs concentration (Fig. 1A). TiO₂-NPs were more aggregated (i.e. higher apparent hydrodynamic diameter) in ultrapure water than in soil solution (126 to 173 nm vs 86 to 153 nm, Fig. 1A) in case of concentrations ranging from 0.625 to 50 mg L⁻¹. At the highest concentrations tested (100 and 125 mg L⁻¹), TiO₂-NPs presented a strong and

similar aggregation with an average apparent diameter of about 160 nm in both ultrapure water and soil solution.

The OP of TiO₂-NPs measured in soil solution and ultrapure water assessed by DCFH oxidation was maximum for the lowest concentration tested (0.625 mg L⁻¹) and decreased progressively with increasing TiO₂-NPs concentration (Fig. 1B). The oxidative activity of TiO₂-NPs was higher in soil solution than in ultrapure water for all concentrations tested except for the 12.5 mg L⁻¹ (Fig. 1B). In soil solution, the OP of TiO₂-NPs ranged from 35 to 0.2 fluorescence units sec⁻¹ μg TiO₂⁻¹ between 0.625 and 125 mg L⁻¹, while in ultrapure water the OP ranged from 5 to 0.02 fluorescence units sec⁻¹ μg⁻¹ TiO₂.

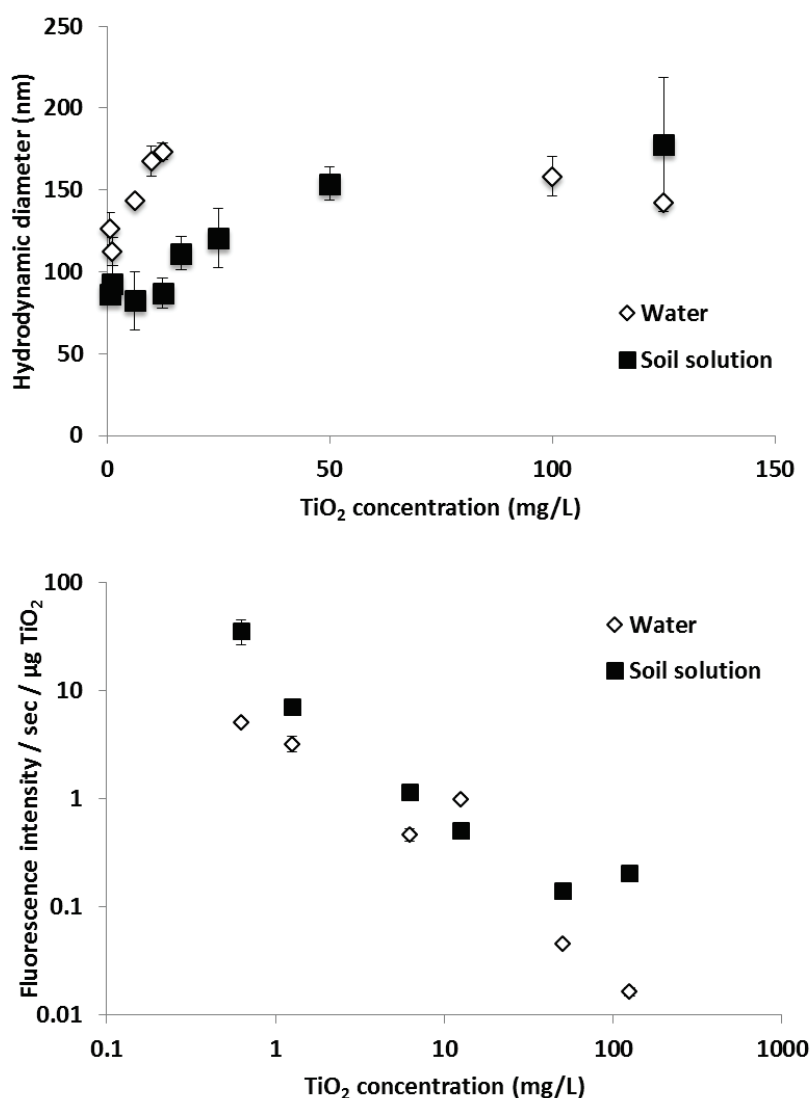


Figure 1 Influence of TiO₂-NPs concentration in ultrapure water or soil solution on A) the aggregation of TiO₂-NPs measured by Dynamic Light Scattering and B) the oxidative potential

(OP) assessed by DCFH oxidation and expressed as Fluorescence intensity (Arbitrary units) per second and per µg of TiO₂-NPs.

Ammonia- and nitrite oxidizer abundance

In this soil, among ammonia-oxidizers, the AOA were dominant over AOB (54 fold higher abundance on average) and among nitrite-oxidizers, the *Nitrospira* genus was dominant (441 fold higher abundance) over *Nitrobacter*.

After 90 days of exposure in microcosms, AOA abundance was similarly reduced (around -40%) by low (Fig. 2A, 0.1 mg kg⁻¹; 0.5 mg kg⁻¹), intermediate (10 mg kg⁻¹) or high TiO₂-NPs concentrations (100 mg kg⁻¹; 500 mg kg⁻¹). The concentrations 0.05, 1 and 50 mg kg⁻¹ lowered AOA abundance (-30 to -35%) but these effects were found not significant because of a high variability of these qPCR data. In contrast, AOB abundance was significantly decreased only by intermediate concentrations of TiO₂-NPs (Fig. 2B, 1 mg kg⁻¹: -32%; 10 mg kg⁻¹: -29%). Low and intermediate concentrations of TiO₂-NPs reduced *Nitrobacter*-like NOB abundance (Fig. 2C, 0.1 mg kg⁻¹: -28%; 0.5 mg kg⁻¹: -32%; 1 mg kg⁻¹: -33%; 10 mg kg⁻¹: -36%). A marginally significant reduction was also observed for a high TiO₂-NPs concentration (100 mg kg⁻¹: -30%). On the contrary, no significant was found on *Nitrospira* abundance whatever the concentration (Fig. 2D).

Table 3 Correlations between nitrifiers abundance (AOA, AOB, *Nitrobacter*-like NOB and *Nitrospira*) on all data (n=36). The R² values are given and significant correlations (*, P<0.05; ***, P<0.001) are presented in bold.

	AOA	AOB	<i>Nitrobacter</i>
AOB	0.09		
<i>Nitrobacter</i>	0.18 *	0.75 ***	
<i>Nitrospira</i>	0.11	0.42 ***	0.36 ***

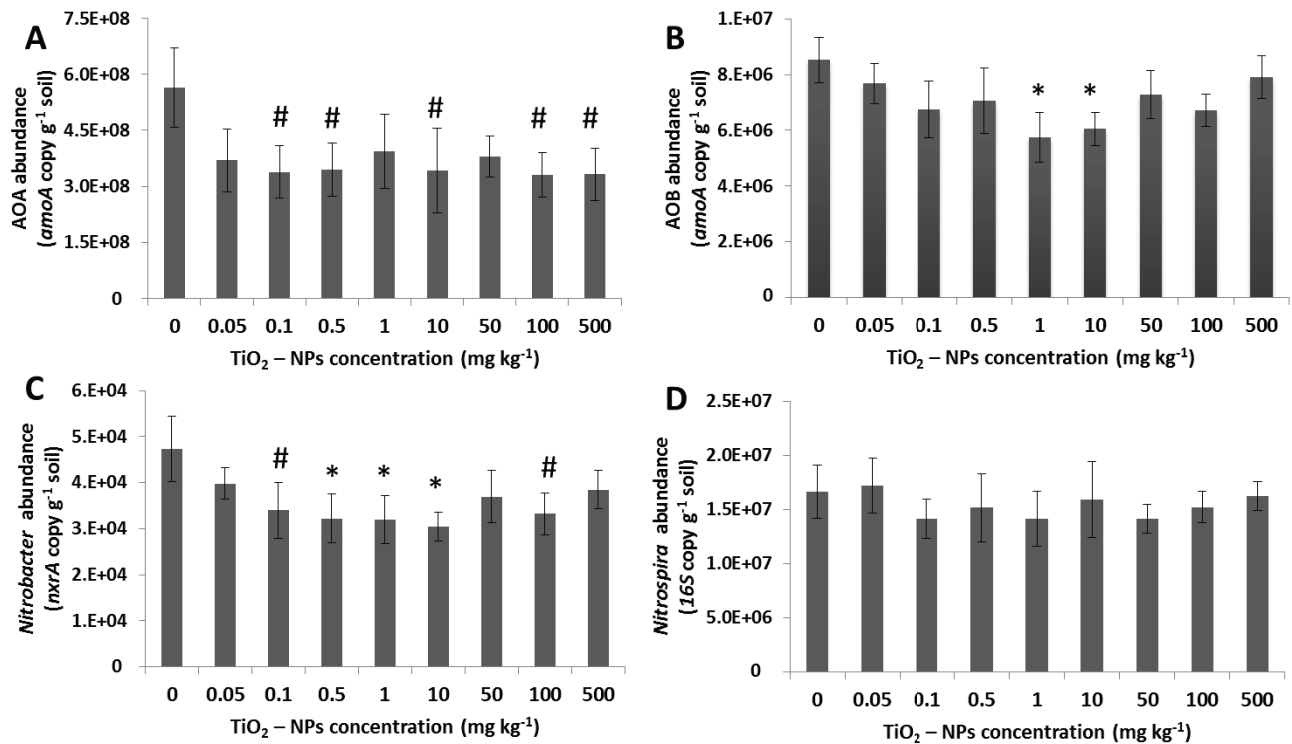


Figure 2 Effect of the different concentrations of TiO₂-NPs after 90 days of exposure on abundances of: A) AOA, B) AOB, C) *Nitrobacter*-like NOB, D) *Nitrospira*. Means and standard errors are presented (n=4). Significant effects and marginally significant effects are indicated (#, $P < 0.1$; *, $P < 0.05$).

The abundance of AOA and AOB was not correlated, whereas the abundance of *Nitrobacter*-like NOB and *Nitrospira* was positively correlated (Table 3). A strong correlation was found between *Nitrobacter*-like NOB and AOB abundance, while *Nitrobacter*-like NOB abundance was weakly correlated to AOA abundance (Table 3). Similarly, *Nitrospira* abundance was positively correlated to AOB abundance, but not AOA abundance (Table 3).

Nitrification activity

After 90 days of exposure in microcosms, soil potential nitrification was significantly decreased by 25% and 26% by the lowest concentration of TiO₂-NPs (Fig. 3) and the two

highest concentrations, respectively (100 mg and 500 mg kg⁻¹). The other concentrations ranging from 0.1 to 50 mg kg⁻¹ had no effect on the nitrification activity.

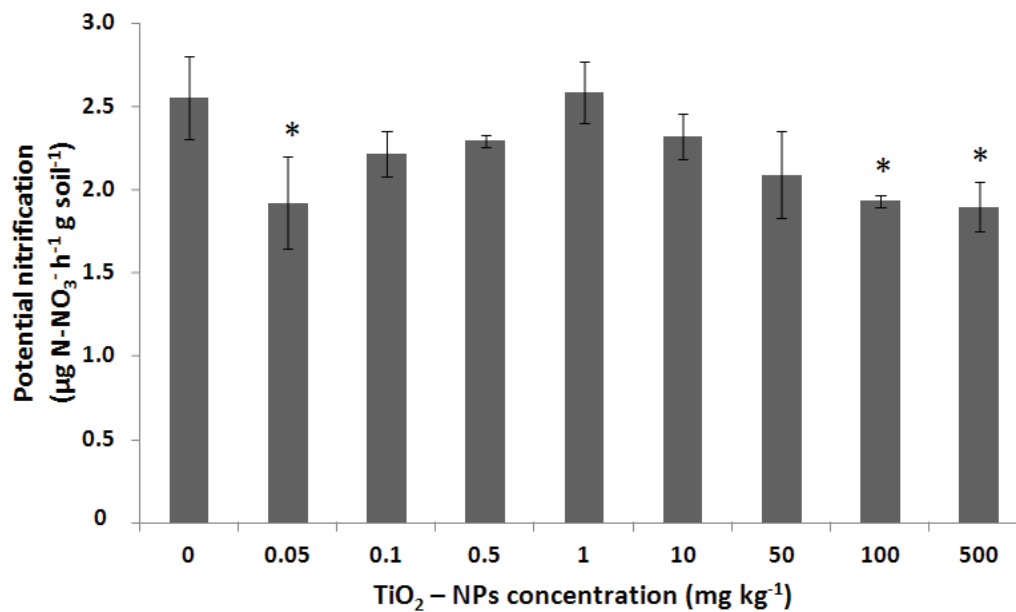


Figure 3 Effect of increasing concentrations of TiO₂-NPs on soil potential nitrification activity measured after 90 days of exposure. Means and standard errors are presented (n=4). Significant effects are indicated (*, $P < 0.05$).

Potential nitrification was not correlated to any of the nitrifier abundance measured (data not shown) when considering the entire data set (n=36). However, a positive correlation was found between nitrification activity and AOA abundance in the samples exhibiting a significantly reduced nitrifying activity (concentrations 0.05, 100 and 500 mg kg⁻¹, Fig. 4).

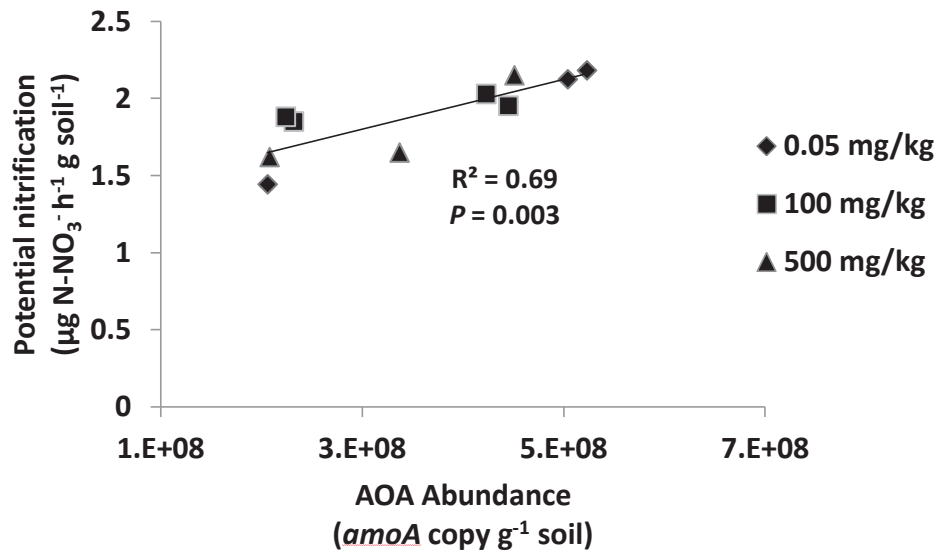


Figure 4 Correlation between potential nitrification and AOA abundance in the samples spiked with 0.05, 100 and 500 mg kg⁻¹.

Path analysis

In order to determine whether the effects of TiO₂-NPs on potential nitrification could be explained by the modification of nitrifier abundance and of the coupling between ammonia- and nitrite-oxidizers, we used a path analysis approach. These path analysis were performed based on a theoretical full model (Fig. 5A), which includes all possible relationships between ammonia-, nitrite-oxidizer abundance and potential nitrification activity. Final models with the best fit were obtained after removing the paths that did not contribute to explain the variations of nitrification or NOB abundance in the model.

When considering the complete data set of the TiO₂-NPs concentrations tested, the path analysis enabled to explain a low part of the variation of the nitrification activity (Fig. 5B, R²=16%). The variables that explained the largest part of potential nitrification variation were AOB and *Nitrobacter* abundance (AOB: path coefficient=-0.54, P=0.06; *Nitrobacter*: path coefficient=0.18, P=0.53). In addition, *Nitrobacter* abundance was mainly controlled by AOB abundance (path coefficient=0.85, P<0.001) and more weakly by AOA abundance (path coefficient=0.17, P=0.03), explaining a total variation of 75% of the *Nitrobacter* data.

We also performed a path analysis based on 3 different subsamples of the data set according to the TiO₂-NPs concentration (i.e low, intermediate or high concentrations, Fig. 5C, 5D and 5E). In the low concentration data set (Fig. 5C), the path analysis indicated that AOA and *Nitrospira* abundance explained 94% of nitrification variation (AOA: path coefficient=0.21, $P=0.008$; *Nitrobacter*: path coefficient=-0.95, $P<0.001$). This model also showed that AOB abundance was an important predictor of *Nitrospira* abundance ($R^2=0.87$, path coefficient=0.93, $P<0.001$).

In the intermediate concentration data set (Fig. 5D), the four nitrifier groups were directly (*Nitrobacter* and *Nitrospira*) or indirectly related (AOA and AOB) to potential nitrification and explained 41% of nitrification variation. AOA abundance influenced *Nitrospira* ($R^2=0.45$, path coefficient=0.67, $P=0.045$) and *Nitrobacter* abundance (path coefficient= 0.25, $P=0.002$). However, AOB abundance was the most important predictor of *Nitrobacter* abundance (path coefficient= 0.88, $P<0.001$).

In the high concentration data set (Fig. 5E), the most important explanatory variable of nitrification activity was AOA abundance (path coefficient=0.70, $P=0.002$) followed by *Nitrobacter* abundance (path coefficient=-0.28, $P=0.22$), explaining 67% of variation in the nitrification data. Furthermore, *Nitrobacter* abundance was mainly controlled by AOB abundance (path coefficient=0.87, $P<0.001$) and more weakly by AOA abundance (path coefficient=-0.25, $P=0.13$), explaining a total variation of 75% of the *Nitrobacter* data.

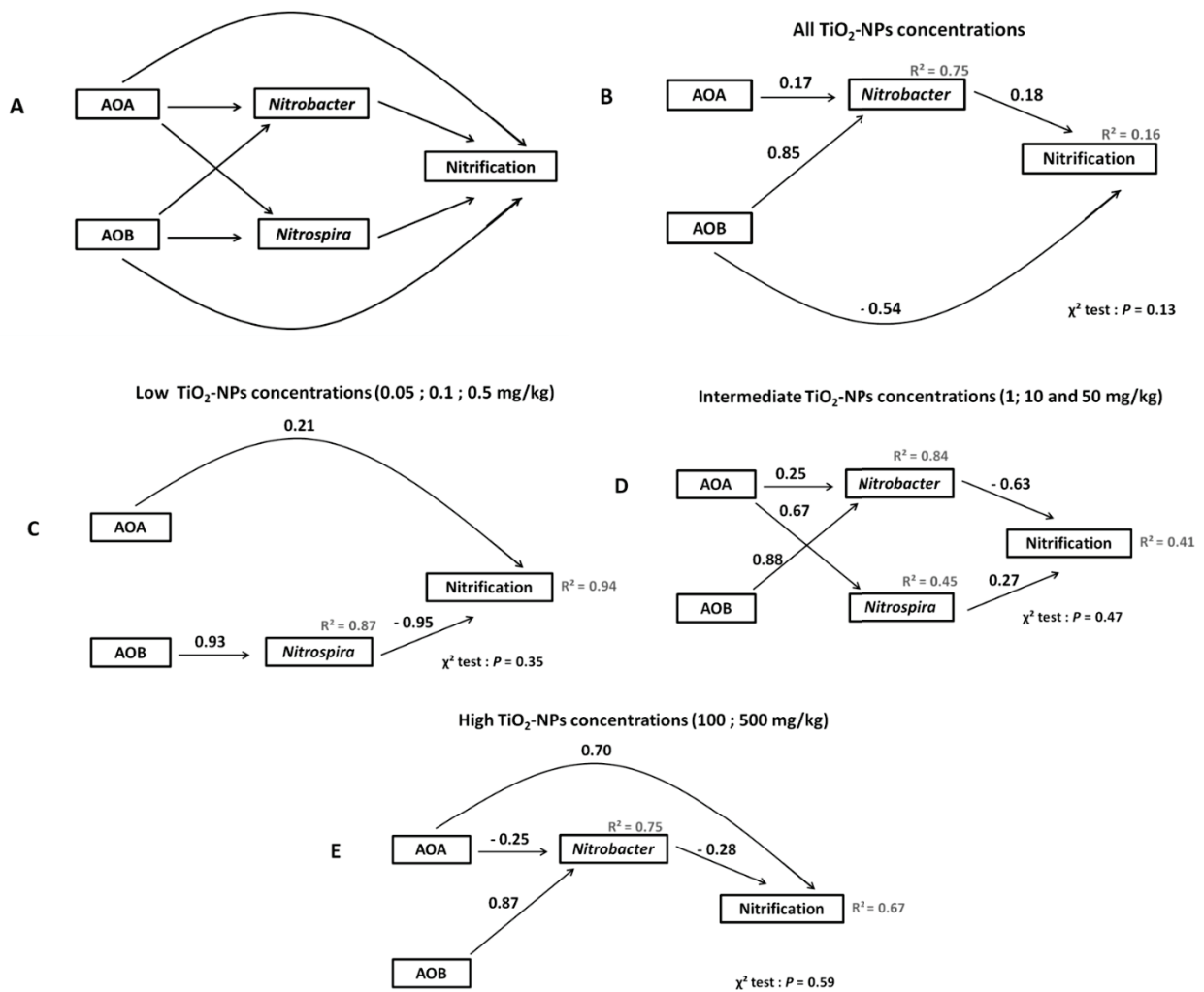


Figure 5 Path diagrams representing the theoretical full model (A) and the final models retained (B, C, D, E) to describe patterns observed in potential nitrification activity. Path diagram based on the data set with B) all TiO₂ concentrations tested, C) only the low concentrations tested (0.05; 0.1; 0.5 mg kg⁻¹), D) only the intermediate concentrations (1, 10, 50 mg kg⁻¹) and E) only the high concentrations tested (100 and 500 mg kg⁻¹). The final models were obtained when the best model-fit (χ² test) was achieved after removing the paths in the model in a stepwise manner. Path coefficients (values indicated next to the arrows) correspond to the standardized coefficients calculated based on the analysis of correlation matrices and indicate by how many standard deviations the effect variable would

change if the causal variable was changed by one standard deviation. The explained variation of the effects variables are given by the R² values (in grey).

DISCUSSION

Influence of the TiO₂-NPs concentration on aggregation and oxidative potential

TiO₂-NPs properties are modified according to the concentration present in water or in soil solution. NPs are known to immediately aggregate when added in aqueous media (Lowry et al. 2012). As already reported, we observed that the aggregation increased with the concentration in both ultrapure water and soil solution, likely due to the higher probability of particle collisions (Maximova and Dahl, 2006; Baalousha, 2009; Jeong et al., 2012). However for the concentrations below 50 mg L⁻¹, TiO₂-NPs were less aggregated in the soil solution than in ultrapure water. This is presumably due to the interactions of NPs with natural organic matter known to modify NPs surface properties and to increase steric repulsion (Baalousha, 2009; Domingos et al., 2009; Thio et al., 2011; Simonin et al., 2015a). From these results, we can assume that the NPs initially prepared in ultrapure water became less aggregated after application in soil and mixing with soil pore water, especially at the lowest concentrations.

The OP of TiO₂-NPs was also affected by their concentration. The increase of TiO₂-NPs concentration resulted in a strong decrease of their OP. These results could be related in part to the simultaneous increase of TiO₂-NPs aggregation with increasing concentration. The reactivity of NPs is generally linked to the available specific surface area of the particles (Hotze et al., 2010a; Lowry et al., 2012). The aggregation induced a decrease of the TiO₂-NPs surface area and consequently a decrease of their OP (Hotze et al., 2010b). TiO₂-NPs oxidative activity was 1.2 to 11 fold higher in soil solution than in ultrapure water and this result might be linked to their lower aggregation in soil solution. However, for the highest concentrations, the aggregation of TiO₂-NPs was similar in ultrapure water and soil solution (100 and 125 mg L⁻¹) but the oxidative activity was still higher in soil solution than in ultrapure water. These results suggest that NPs aggregation is not the only factor involved and that physicochemical properties of the soil solution might enhance the amount of ROS produced from TiO₂-NPs. In particular, metals, such as iron, present in the soil solution could

be a secondary source of ROS that depend on the dissolved organic carbon present in the solution (Gandois et al., 2010; Charrier and Anastasio, 2012).

In contrast with highly soluble pollutants (e.g. heavy metals, pesticides), TiO₂-NPs properties involved in the bioavailability and the toxicity of these emerging pollutants, such as aggregation and ROS production, are highly modified according to the concentration. This finding highlights the necessity to characterize NPs properties for each concentration used in conditions close to those encountered in soil.

Contrasted responses of ammonia-oxidizers and nitrite-oxidizers to TiO₂-NPs

The four microbial groups involved in soil nitrification responded differently to TiO₂-NPs contamination after 90 days of exposure. In this soil AOA appeared to be the most affected by TiO₂-NPs, while *Nitrospira* seemed insensitive to this contamination. AOB and *Nitrobacter* abundance exhibited similar responses illustrated by a strong positive correlation between these two groups. None of the microbial abundance measured exhibited a linear dose-response relationship in the range of the concentrations tested. Indeed, concentrations that differed by a factor of 5000 for AOA (0.1 and 500 mg kg⁻¹) or 1000 for *Nitrobacter* (0.1 and 100 mg kg⁻¹) caused a similar decrease of abundance. Furthermore, AOB abundance was reduced only at 2 intermediate concentrations (1 and 10 mg kg⁻¹). These results are not surprising as TiO₂-NPs properties are modified according to the concentration. Unlike soluble chemicals, dose-response relationships do not seem to be workable in the case of NPs in soil.

In the available literature, this is the first report of negative effects of TiO₂-NPs on soil microbial communities at such very low concentrations (< 1 mg kg⁻¹). These effects might be explained by the low aggregation and high reactivity of TiO₂-NPs likely conferring a higher toxicity and bioavailability at these low concentrations (Choi and Hu, 2009; Simon-Deckers et al., 2009; Menard et al., 2011). The effects observed at intermediate and high concentrations, despite their presumed lower reactivity and bioavailability, might be linked to a greater probability of interaction between TiO₂-NPs and soil microorganisms.

The higher sensitivity of AOA towards TiO₂-NPs was surprising as it is commonly agreed that these microorganisms are more tolerant to chronic stresses due to the specific chemical

structure of their membrane lipids which is exceptional in the microbial world (Schleper et al., 2005; Valentine, 2007). Indeed, soil AOA have been reported to be more tolerant to heavy metals than AOB (Li et al., 2009; Ollivier et al., 2012; Subrahmanyam et al., 2014), but not in all cases (Mertens et al., 2009). However, the current lack of knowledge on soil AOA physiology and ecology does not allow to propose any hypothesis for their higher sensitivity to TiO₂-NPs.

AOB and *Nitrobacter* abundance was significantly lowered at the intermediate TiO₂-NPs concentrations tested. Previous studies conducted with activated sludges, also reported a decrease of AOB and NOB (*Nitrobacter* and *Nitrospira*) abundance associated to a high reduction of ammonia- and nitrite-oxidation activities in presence of TiO₂-NPs or Ag-NPs (Liang et al., 2010; Zheng et al., 2011). However, in our study *Nitrospira* abundance was not modified whatever the concentration of TiO₂-NPs. Among soil nitrite oxidizers, *Nitrobacter* and *Nitrospira* have different ecological requirements, *Nitrobacter* being adapted to high nitrite and oxygen environments, while *Nitrospira* are adapted to low nitrite and oxygen conditions (Schramm et al., 1999; Attard et al., 2010; Wertz et al., 2012; Nowka et al., 2015; Simonin et al., 2015b). In this study, the decreases of AOB and AOA abundance and activity may have resulted in lower nitrite availability and consequently conditions favorable to *Nitrospira* compared to *Nitrobacter* (Ollivier et al., 2013).

These contrasted responses observed among ammonia- and nitrite-oxidizers according to the NPs concentrations tested suggest a different sensitivity of the four microbial groups to TiO₂-NPs and especially to their oxidative stress. Consequently, under this disturbance some modifications in the functional players of ammonia and nitrite-oxidation could be expected depending on the concentration.

Consequences on nitrification activity: a path analysis approach

As the abundance of microbial groups involved in the two steps of the nitrification was variably altered by TiO₂-NPs contamination, we used a path analysis to decipher how changes in ammonia- and nitrite-oxidizer abundance have impacted soil nitrification activity.

Interestingly, potential nitrification activity was found to be reduced only with the lowest and the highest concentrations, whereas significant decreases in nitrifier abundance were observed also in intermediate TiO₂-NPs concentrations. The path analysis encompassing all TiO₂-NPs concentrations tested did not allow to explain satisfactorily the changes in nitrification activity, probably because the links between ammonia-oxidizer abundance, nitrite-oxidizer abundance and nitrification were concentration-dependent. This is why the path analysis were further performed separately with low, intermediate and high NPs concentrations data sets. With this approach a significant part of the nitrification activity variation was explained by the models. The functional players of nitrification and the coupling between ammonia- and nitrite-oxidizers were different in the 3 path analysis (i.e low, intermediate and high concentrations). These results suggest a surprising flexibility of nitrification functioning supported by a functional redundancy among ammonia-oxidizers and nitrite-oxidizers (Wertz et al., 2007; Mertens et al., 2009). However, the significant decreases of potential nitrification induced by 3 TiO₂-NPs concentrations (0.05; 100 and 500 mg kg⁻¹) suggest that the nitrifier functional redundancy does not always allow to maintain nitrification efficiency.

In the low and high TiO₂-NPs concentrations path analyses, AOA abundance was directly linked to nitrification, suggesting their important functional role in ammonia-oxidation in this soil. Consistently, AOA abundance was on average 54 fold higher in this soil than AOB abundance. The direct path between AOA abundance and nitrification may further indicate that ammonia-oxidation can be the limiting step of nitrification as often reported (Kowalchuk and Stephen, 2001). Thus we can assume that the observed significant decreases of potential nitrification could be partially related to the decreases of AOA abundance. This assumption is supported by the positive correlation between AOA and potential nitrification (Fig. 4)

The variation of ammonia-oxidizer abundance enabled to explain a great part of NOB abundance variation (*Nitrobacter* and/or *Nitrospira* according to the TiO₂-NPs concentrations) highlighting the strong coupling between ammonia- and nitrite-oxidizers in soil (Ollivier et al., 2013; Simonin et al., 2015b). Therefore, the shifts in NOB community and the altered coupling between ammonia- and nitrite-oxidizers might also be responsible for the lower nitrification efficiency in some treatments. In particular, culture studies have

shown that AOA and *Nitrospira* have lower specific cell activity than AOB and *Nitrobacter*, respectively (Schramm et al., 1999; Prosser and Nicol, 2012; Nowka et al., 2015).

As also reported by Petersen et al. (2012), this study shows that functional gene abundance is a good predictor of potential rates that integrates recent environmental history and recent process activity. Thereby the path analysis approach permitted to understand how a disturbance altered the coupling between ammonia- and nitrite-oxidizers and the subsequent effects on potential nitrification activity. However, some variations of nitrification activity remain unexplained in the path analysis. Approaches targeting functional gene transcripts and/or DNA stable isotope probing could help identifying active nitrifiers in this soil and deciphering the contributions of AOB, AOA, *Nitrobacter* and *Nitrospira* communities to nitrification activity (Xia et al., 2011; Ruyters et al., 2013). The improvement of the path analysis could also be achieved by considering the effects of TiO₂-NPs on nitrifier diversity and community structure in order to include specific nitrifier lineages involved in nitrification and/or sensitive to TiO₂-NPs stress in the model (Cantarel et al., 2012).

Conclusions

In this soil, the nitrification was similarly reduced by a very low TiO₂-NPs concentration (0.05 mg kg⁻¹) and high concentrations (100 and 500 mg kg⁻¹). Changes in nitrification activity were explained by the alteration of the coupling between ammonia- and nitrite-oxidizers due to the different sensitivity of these microbial groups to TiO₂-NPs contamination according to the concentration.

The absence of classical dose-response relationships on nitrifier abundance was related to the modification of TiO₂-NPs properties depending on the concentration tested. TiO₂-NPs exhibited the largest toxic potential in soil solution at the lowest concentrations, due to low aggregation and high oxidative potential under these conditions. This emphasized that classical approaches for risk assessment based on dose-response toxicological tests are not pertinent for NPs in soil.

This work highlights that TiO₂-NPs could be detrimental for a key process of the N cycling and raises concern about the potential cascading effects on other essential processes for ecosystem services delivery.

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CONFLICT OF INTEREST

The authors declare to have no conflict of interest.

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4. Comparaison des effets de contaminations aiguë et chronique sur le transport et la toxicité du TiO₂ sur la nitrification

a. Article 6 : Présentation générale de l'étude et synthèse des principaux résultats

Nos travaux ont permis de montrer que la toxicité des NPs peut s'exprimer après une durée de trois mois dans des microcosmes de sols et que les communautés microbiennes affectées sont peu résilientes à cette perturbation. Il s'agissait, dans toutes les expériences, de contaminations aiguës aux TiO₂-NPs (i.e. 1 seule contamination), or il est plus vraisemblable que les sols soient exposés à ces contaminants de manière chronique à court ou long terme. Par exemple, l'irrigation ou l'utilisation de nanofertilisants ou nanopesticides peuvent être renouvelés plusieurs fois la même année entraînant une exposition chronique des sols agricoles. L'absence de résilience et les effets apparaissant à long terme dans nos précédentes expérimentations suscitent des interrogations quant à la réponse des microorganismes exposés de façon chronique aux TiO₂-NPs. En effet, on peut faire l'hypothèse que les effets observés sur des communautés microbiennes déjà exposées à cette perturbation mais non résistantes sera amplifié lorsqu'une nouvelle contamination aura lieu. Il est également possible que des effets négatifs n'apparaissent qu'après un apport chronique. Ces aspects n'ayant pas fait l'objet d'études répertoriées dans la littérature, il nous a paru nécessaire de mettre en place des dispositifs permettant d'étudier à la fois les conséquences de contaminations aiguë et chronique sur les microorganismes du sol. L'utilisation de microcosmes de sol ne permet pas de réaliser des apports multiples de NPs, lorsqu'elles sont apportées en suspension, sans modifier le statut hydrique du milieu. Par ailleurs, les microcosmes de sol constituent un système statique dans lequel les transferts verticaux sont difficilement pris en compte.

Dans des situations réalistes de contamination du sol par les NPs, ces dernières seront apportées en surface et pourront être entraînées en profondeur par les mouvements d'eau, par exemple suite à un arrosage ou une pluie. Dans ces conditions, on peut penser que la distribution des NPs dans les sols sera hétérogène et on peut s'attendre à un gradient de concentration du polluant en fonction de la profondeur, selon leurs capacités de transport dans le sol considéré. L'absence de relation dose-réponse linéaire associée aux TiO₂-NPs rend difficile la prédiction de l'effet d'une contamination ayant une distribution hétérogène

dans le sol, car de fortes concentrations peuvent avoir des effets similaires à ceux de faibles concentrations (Article 5).

Nous proposons donc d'utiliser un système dynamique basé sur des colonnes de sol comme nous l'avons utilisé pour les expériences de transport (Article 2), afin d'apprécier les effets d'une exposition plus réaliste par un apport en surface. Ce dispositif permet de réaliser à la fois des contaminations aiguës et chroniques et d'étudier en parallèle le transport du contaminant et son impact sur les communautés microbiennes à différentes profondeurs du sol. Dans notre expérimentation, nous avons réalisé 3 scénarios d'exposition permettant d'obtenir une même concentration totale finale en TiO₂-NPs (372 mg kg⁻¹) apportée dans le sol limono-argileux :

- Scénario 1 : Contamination aiguë= 1 apport de TiO₂-NPs à 50 mg L⁻¹
- Scénario 2 : Contamination chronique= 2 apports de TiO₂-NPs à 25 mg L⁻¹
- Scénario 3 : Contamination chronique= 3 apports de TiO₂-NPs à 16.7 mg L⁻¹

Pour les colonnes recevant plusieurs apports de TiO₂-NPs, les différentes contaminations ont été réalisées avec un intervalle de 15 jours d'incubation entre chaque. Durant chaque apport, les effluents des colonnes ont été collectés afin d'étudier le transport du TiO₂.

Pour chacun de ces scénarios, après la dernière exposition, le sol des colonnes a été récupéré, découpé en différentes couches en fonction de la profondeur (0-2 cm ; 2-4 cm ; 4-6 cm ; 6-8 cm) puis incubé dans des microcosmes pendant 60 jours afin d'évaluer l'impact des NPs sur les communautés microbiennes (Figure 17). L'indicateur écologique choisis a été la nitrification en mesurant les effets sur l'activité nitrifiante et l'abondance des AOA et AOB.

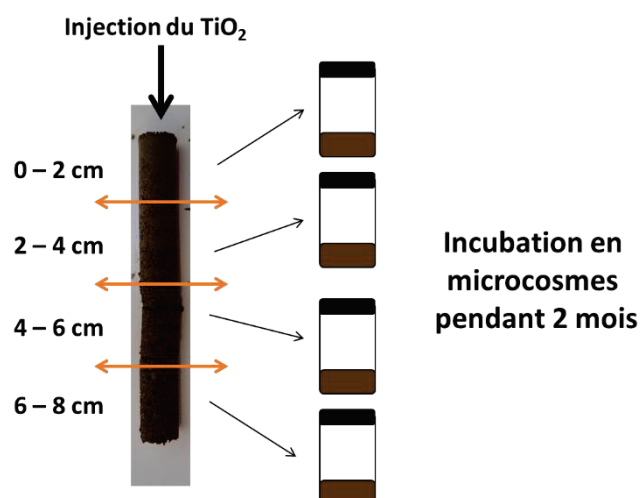


Figure 17: Découpage d'une colonne de sol en 4 profondeurs avant son incubation en microcosmes.

Chapitre 3 : Impact du TiO₂ sur le cycle de l'azote : exemple d'un sol limoneux-argileux

En accord avec l'article 2, nos résultats montrent que le transport du TiO₂ est très faible et en plus qu'il dépend de la concentration appliquée lors d'une contamination aiguë (i.e. 1^{er} apport). La mobilité du TiO₂ augmente lorsque la concentration utilisée diminue (bilan de masse TiO₂ : 50 mg L⁻¹ = 2.2% ; 25 mg L⁻¹ = 5% ; 16.7 mg L⁻¹ = 11%). Cela peut être expliqué par le fait que l'agrégation des TiO₂-NPs diminue aux plus faibles concentrations, ce qui favorise leur transport dans la porosité du sol. Lors de la seconde contamination, nous avons constaté que la mobilité du TiO₂ était fortement réduite comparée au premier apport, quelle que soit la concentration testée (25 et 16.7 mg L⁻¹). Pour les colonnes recevant un 3^{ème} apport, le transport du polluant était négligeable car 99.8% du TiO₂ amené était retenu dans le sol. La concentration finale retenue dans le sol a été calculée à partir des bilans de masse obtenus. Nous avons observé que la concentration en TiO₂ la plus forte était observée dans les colonnes soumises à une contamination aiguë (364 mg kg⁻¹) et la plus faible dans celles exposées 3 fois aux TiO₂-NPs (328 mg kg⁻¹).

Concernant l'évaluation de l'impact sur la nitrification dans les 3 scénarios d'exposition, aucun effet négatif n'a été observé dans les sols soumis à 1 ou 2 apports de TiO₂-NPs. En revanche, une diminution significative de l'activité nitrifiante et de l'abondance des AOA et des AOB a été constatée après 3 apports de TiO₂-NPs sur l'ensemble de la colonne (i.e. moyenne des 4 profondeurs). Un effet significatif de la profondeur a été uniquement observé sur l'activité nitrifiante pour laquelle une diminution plus importante a été notée dans les 2 premiers centimètres du sol, zone probable d'accumulation préférentielle de NPs comme le prédit la théorie de la filtration des colloïdes (e.g. Vitorge 2010, Vitorge *et al.*, 2014).

En conclusion, ces travaux montrent qu'il est fondamental de comparer les effets de contaminations aiguës et chroniques en conditions réalistes d'exposition. En effet dans ces conditions simulant un apport en surface, une contamination aiguë à forte concentration n'a pas eu d'effet alors que 3 contaminations résultant à une concentration 10% plus faible ont eu des effets délétères sur la nitrification. Il serait nécessaire de considérer la résistance et la résilience des communautés microbiennes lors d'expositions chroniques en étudiant les effets après chaque exposition et avant la suivante pour comprendre et prédire l'effet de contaminations répétées.

Dans cette expérimentation, nous avons également mis en évidence qu'à partir de la seconde exposition, les TiO₂-NPs étaient peu mobiles dans le sol, sans doute suite au

colmatage partiel des pores situés dans les premiers centimètres du sol, qui ne permettent alors plus le transport de ces contaminants. Toutefois, pour le vérifier nous n'avons pas pu mesurer la concentration en TiO₂-NPs aux différentes profondeurs du sol en raison de la très forte concentration naturelle du sol en titane (4.52 g kg⁻¹) comparée aux concentrations apportées.

b. « Transport of TiO₂ nanoparticles and toxicity on microbial communities under acute and chronic exposures in soil columns»

L'article 6 intitulé «Transport of TiO₂ nanoparticles and toxicity on microbial communities under acute and chronic exposures in soil columns» est en préparation en vue d'une soumission dans la revue *Environmental Science & Technology*.

Transport of TiO₂ nanoparticles and toxicity on microbial communities under acute and chronic exposures in soil columns

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ABSTRACT

Soils are being extensively exposed to nanoparticles (NPs) due to their increasing use in many commercial products. The adverse effect of NPs on soil microorganisms has been reported in several ecotoxicological studies with various experimental setups, such as microcosms or field lysimeters. Most of these studies were based on acute exposures to NPs although chronic exposures are more likely to occur. Contrarily to single acute contaminations, the evaluation of soil exposure to chronic NPs contaminations requires the use of specific setups. Using a soil column approach, we compared the influence of acute and chronic contaminations on the transport of TiO₂-NPs through soil. The effect of these different exposure scenarios on the abundance and activity of soil nitrifying microbial community after 2 months of incubation was evaluated. Soil columns were contaminated using 3 exposure scenarios (1, 2 or 3 TiO₂-NPs applications) resulting in the same TiO₂-NPs final concentration applied in columns of a silty-clay soil. The transport of TiO₂-NPs was very low in this soil under both acute and chronic exposures and was influenced by NPs injection concentration during the first application of each scenario. TiO₂-NPs mobility increased at lower concentration wherein TiO₂-NPs were less aggregated in the soil solution compared to the highest concentration. Significant decreases of nitrification activity and ammonia-oxidizing archaeal and bacterial abundances were observed only in chronic exposure condition, in the 3 TiO₂-NPs applications scenario.

Thus, this study highlights that under chronic exposure, the transport of TiO₂-NPs to deeper soil layers and groundwater is likely limited and that a chronic contamination was more damaging for the microbiological functioning than an acute exposure. This work suggests that more studies on the fate and impact of chronic NPs contaminations are clearly needed as they appear, contrarily to acute exposure, as the more favourable situations for NPs impact in soil

Key-words: TiO₂ Nanomaterials, Chronic and acute exposure, transport, Microbial Ecotoxicology, Nitrification.

INTRODUCTION

In recent years, a significant number of studies highlighted deleterious effects of metal-oxide nanoparticles (NPs) on soil (micro)organisms and consequently on soil functioning and fertility (Dinesh *et al.*, 2012; Pan and Xing, 2012; Simonin and Richaume, 2015). In most of these studies, soils were exposed to acute homogeneous NPs contaminations in microcosms (Simonin and Richaume, 2015). However, NPs are more likely to be released on soil surface through repeated applications of sewage sludge, irrigation or the use of nanofertilizers and nanopesticides (Brar *et al.*, 2010; Liu and Lal, 2015; Servin *et al.*, 2015) and are then leached in depth into soil porosity (Cornelis *et al.*, 2014). Thereby in natural soils, NPs are heterogeneously distributed, especially along the vertical soil profile (Navarro *et al.*, 2011). This heterogeneous distribution of NPs is largely controlled by the soil physicochemical properties that influence their mobility in agreement with the colloidal filtration theory (Fang *et al.*, 2009).

Furthermore although soils are mostly chronically exposed to NPs, most transport and ecotoxicological studies generally consider a single and high NPs concentration as a chronic exposure. To date, only few studies have actually performed experiments with repeated soil exposures to NPs (Hooper *et al.*, 2011). The lack of such studies and consequently of knowledge on NPs chronic toxicity in soil is mainly due to the difficulty to perform chronic exposures in microcosm experiments. In microcosms, soil moisture must be kept constant all along the soil incubation and repeated aqueous applications of NPs for simulating chronic exposures would induce moisture stresses in addition to the NPs toxicity, thus limiting the validity of such approaches. Therefore, several aspects remain to be investigated to understand the fate and toxicity of NPs. Specifically, it is necessary to use more realistic experimental conditions of contamination simulating chronic exposure with the application of NPs on soil surface, in order to consider their heterogeneous distribution according to soil depth.

The aim of this work was to compare the effect of acute and chronic contaminations on (i) the transport of titanium dioxide NPs (TiO₂-NPs) through a soil profile and (ii) their toxicity on soil microbial community by focusing on the key microbial process of nitrification involved in the nitrogen cycle. TiO₂-NPs were chosen because they are the most produced

NPs and it is predicted that they would represent 50% of all NPs retrieved in soil (Keller *et al.*, 2013; Sun *et al.*, 2014).

The experimental design was based on soil columns and enabled the study of the fate and impact of both acute and chronic TiO₂-NPs exposures in soil under more realistic conditions of exposure than those usually carried out using microcosms. Soil columns were contaminated using 3 exposure scenarios (1, 2 or 3 contaminations) resulting in the same total TiO₂-NPs concentrations applied in soil columns (372 mg NPs kg⁻¹ dry soil). After the last NPs application, four layers of the contaminated soil columns corresponding to different depths were incubated separately into glass flasks for 2 months to study the NPs impact on the nitrification process. The potential nitrification activity and the abundance of ammonia-oxidizing archaea and bacteria (AOA and AOB respectively) involved in the first step of nitrification were measured and interpreted in the light of NPs physicochemical properties and mobility in soil.

MATERIALS AND METHODS

Soil

The upper 20 cm layer of a silty-clay soil (Cambisol, WRB, 2006) was collected in Commarin (Côte d'Or, France) under a permanent pasture. After collection, roots and plant litter were manually removed. The soil was sieved at 2 mm and homogenized before storage at 4°C. The main soil characteristics were: sand 10%; loam 51%; clay 39%; pH 7.7; OM 7.9%; CEC 20.1 cmol kg⁻¹; water holding capacity (WHC) 51%; ionic strength 1.37 mM.

TiO₂ Nanoparticles

Titanium dioxide nanoparticles (TiO₂-NPs) were provided by Sigma Aldrich (St Louis, USA) (mixture of anatase (80%) and rutile (20%) crystal structure) with at least 99.5% purity. TiO₂-NPs mean particle size was 28.7 nm in powder as measured with a ZEISS Ultra 55 scanning electron microscopy–field emission gun (SEM-FEG) and energy dispersive spectroscopy (EDS) with a SDD detector (BRUKERAXS-30 mm²).

The aggregated size and zeta potential of the TiO₂-NPs were characterized using Dynamic Light Scattering (DLS) in the soil solution used as background solution for the column experiments. Soil solution was prepared following the protocol described by Simonin *et al.* (2015). The measurements were performed in the different suspensions of TiO₂-NPs (50, 25 and 16.7 mg L⁻¹), after dispersion by ultrasonication for 5 minutes to ensure suspensions homogeneity. For each sample, the mean of 3 measurements was recorded.

Exposure of soil columns to acute and chronic applications of TiO₂-NPs

Three TiO₂-NPs suspensions were prepared in soil solution in order to obtain a final total concentration of 372 mg kg⁻¹ dry soil in the soil columns, whatever the number of exposures with 50, 25 and 16.7 mg L⁻¹ for 1, 2 and 3 exposures, respectively.

The soil exposures to NPs were performed in 1 x 10 cm small glass columns (C10/10, GE Healthcare), homogeneously packed with 8 cm of wet soil, at 28°C in the dark. Flow adaptors (AC 10, GE Healthcare) were adjusted on the top of the columns to ensure constant and similar soil heights during experiments. At the beginning of the experiment, the columns were saturated and leached bottom-up with 100 mL of soil solution. Then, the background solution was replaced by the TiO₂-NPs suspension that was injected for 5 pore volumes (PV) (i.e. 6.7 ml) at a flow rate of 0.1 mL min⁻¹. Then, 10 PV of the soil solution free of TiO₂-NPs were injected.

The acute exposure experiment consisted in the injection of 5 PV of 50 mg L⁻¹ TiO₂-NPs into 6 saturated soil columns. Six control columns received the same PV of soil solution free of TiO₂-NPs.

Chronic exposures experiments consisted in two or three applications of TiO₂-NPs (25 and 16.7 mg L⁻¹ respectively) in 6 columns for each treatment. Six additional control columns were realized for each type of exposure. The different applications were separated by a 15 day delay during which columns were maintained horizontally at 28°C in the dark at constant moisture.

Transport of TiO₂-NPs

In each exposure treatment, the effluents from a randomly chosen column were sampled in 15 mL centrifuge tubes every 10 minutes (~1 mL) using a fraction collector (Gilson, Minipulse 3). In order to establish the TiO₂ breakthrough curves, titanium concentrations in the spiking suspensions (C₀) and in the effluents (C) were determined (C/C₀) using a microwave assisted (Novawave, SCP Science) strong acid extraction (hydrofluoric acid + nitric acid). Titanium concentrations were measured by inductively coupled plasma - optical emission spectrophotometer (ICP-OES; Varian 700-ES, Varian Inc. Scientific Instruments, Palo Alto, USA). Control columns leached with NPs-free soil solution enabled to determine the background concentration of titanium in the effluents. In order to ensure the quality of our titanium measurements, a certified reference material was measured along with our samples.

Consistently with Nickel *et al.*, (2015), the TiO₂-NPs concentration retained in the different soil layers after contamination could not be measured due to the high titanium content of the soil (4.52 g kg⁻¹) and the low TiO₂-NPs input used (372 mg kg⁻¹) compared to this background content.

Impact of TiO₂-NPs on nitrification process according to the soil depth

The impact of TiO₂-NPs on nitrification and ammonia-oxidizer abundance in function of soil depth was studied after the different exposure scenarios. After the last TiO₂-NPs exposure, soil columns were cut into four layers (0–2 cm, 2–4 cm, 4–6 cm and 6–8 cm) according to the depth from the inlet and placed into 60 ml glass flasks. This experimental design resulted in 144 microcosms incubated for 2 months at 28°C in the dark at constant moisture: 6 types of column treatments (3 exposure scenarios + 3 controls) x 4 depths x 6 replicates. At the end of the incubation time, 1.5 g of soil (eq dw) were immediately used for the measurements of potential nitrification and 0.5 g of soil were stored at -20°C before DNA extraction.

DNA extraction and quantification of ammonia-oxidizer abundance

DNA was extracted from 0.5 g of frozen soil using the Power Soil™ DNA Isolation Kit (MO BIO laboratories, Carlsbad, CA, USA), following the manufacturer's instructions and then DNA concentrations were determined using the Qubit dsDNA BR Assay (Invitrogen).

The abundance of the ammonia-oxidizers (AOA and AOB) was measured by quantitative PCR using a Lightcycler 480 (Roche Diagnostics, Meylan, France) and the primers and thermal cycling conditions used are described in Table 1.

The *amoA* AOA and AOB quantification were performed in a final reaction volume of 20 µL and contained (final concentrations) 0.5 µM of each primer for the bacterial *amoA* or 0.75µM of CrenamoA616r and 1µM of CrenamoA23f for the archaeal *amoA*, 2% bovine serum albumin (BSA), 1X of QuantiTect SybrGreen PCR Master Mix (Qiagen, Courtaboeuf, France) and 10 ng of soil DNA extract or 10⁷- 10² gene copies number of an in-house plasmid containing cloned bacterial (*Nitrosomonas europaea*, GenBank accession number:L08050) and archaeal (54d9 fosmid fragment, (Treusch *et al.*, 2005)) *amoA* genes. Melting curves analysis confirmed the specificity of amplification of the four genes.

Table 1 PCR primers and thermal cycling conditions used for quantification of ammonia- and nitrite-oxidizer abundances

Primers	Target gene	Reference	Thermal conditions
CrenamoA23f	AOA <i>amoA</i>	Tourna <i>et al.</i> , 2008	15 min at 95°C, 45 cycles (45 s at 94°C,
CrenamoA616r	AOA <i>amoA</i>	Tourna <i>et al.</i> , 2008	45 s at 55°C and 45 s at 72°C)
amoA-1F	AOB <i>amoA</i>	Rotthauwe <i>et al.</i> , 1997	15 min at 95°C, 45 cycles (30 s at 95°C,
amoA-2R	AOB <i>amoA</i>	Rotthauwe <i>et al.</i> , 1997	45 s at 54°C, 45 s at 72°C and 15 s at 80°C)

Nitrification activity

Potential nitrification activity was determined according to the protocol described by Dassonville *et al.*, (2011). Sub-samples of fresh soil (1.5 g equivalent dry soil) were incubated with 3 ml of a solution of (NH₄)₂SO₄ in order to reach 50 µg N-NH₄⁺ g⁻¹ dry soil. Distilled water was added in each sample to achieve 12 ml of total liquid volume in plasma flasks. The flasks were sealed with Parafilm® and incubated at 28°C under 180 rpm constant agitation. During

the incubation, 1 ml of soil slurry was sampled at 2h, 4h, 6h, 8h and 10h, filtered (0.2 µm pore size) and transferred in vials stored at -20°C. The analysis of NO₃⁻ concentrations was performed by ionic chromatography (DX120, Dionex, Salt Lake City, USA) equipped with a 4 mm×250 mm column (IonPac AS9 HC). Potential nitrification was expressed as µg N-NO₃⁻ h⁻¹ g⁻¹ dry soil.

Statistical analysis

All results are presented as means (± standard error). An analysis of variance (ANOVA) and *post-hoc* Tukey HSD were performed to test the effect of TiO₂-NPs exposures and soil depth on the potential nitrification and ammonia-oxidizer abundance for each type of exposure scenario separately. Data were log-transformed prior to analysis when necessary, to ensure conformity with the assumptions of normality and homogeneity of variances. T-tests were conducted to compare the aggregation and zeta potential of TiO₂-NPs in the different spiking suspensions. All statistical analyses were carried out using R statistical software 2.13.2 (R Core Team, 2015).

RESULTS

Characteristics of TiO₂-NPs in the spiking soil solutions

TiO₂-NPs were characterized in the different suspensions used to contaminate the soil columns for the 3 exposure scenarios. The aggregation of TiO₂-NPs increased with concentration (Table 2). In the lowest concentration applied 3 times in the soil column (16.7 mg L⁻¹), TiO₂-NPs hydrodynamic diameter was 111.2 nm in average, while in the highest concentration used for the single exposure (50 mg L⁻¹), TiO₂-NPs hydrodynamic diameter was 153.9 nm.

Surface charge of TiO₂-NPs assessed through zeta potential values were relatively close in the 3 spiking suspensions (-15.4 to -16.6 mV; Table 2), even if it was significantly lower in the 25 mg L⁻¹ suspension.

Table 2 Size and zeta potential of TiO₂-NPs in soil solution for the 3 tested concentrations. Means and standard errors are presented (n=3). Values labeled with the same letter do not differ at $P < 0.05$.

TiO ₂ injection concentration	Hydrodynamic diameter (nm)	Zeta potential (mV)
50 mg L ⁻¹	153.9 ± 10 b	-16.6 ± 0.2 a
25 mg L ⁻¹	120.8 ± 18 ab	-15.4 ± 0.1 b
16.7 mg L ⁻¹	111.2 ± 10 a	-16.4 ± 0.3 ab

Effect of the concentration and the number of applications on TiO₂-NPs mobility in soil

The experimental design enabled to study the influence of TiO₂-NPs concentration on their transport through soil after acute contaminations at 16.7, 25 or 50 mg L⁻¹ (first application) and after chronic exposures consisting in 2 or 3 NPs applications (Table 3).

The mobility of TiO₂-NPs decreased when increasing TiO₂-NPs concentration (Table 3). At an injection concentration of 16.7 mg L⁻¹, 10.66% of injected TiO₂-NPs were recovered in the effluents, whereas only 4.95 and 2.19% were recovered at 25 and 50 mg L⁻¹, respectively.

In the columns receiving two applications, the mobility of TiO₂-NPs was greatly reduced compared to the 1st application (Table 3). Only 0.9% of the added TiO₂-NPs were recovered in the column effluents for both concentrations (Table 3). For the 3rd NPs application at 16.7 mg L⁻¹, the mobility of TiO₂-NPs was almost negligible (0.2%; Table 3).

We calculated the final TiO₂-NPs concentration retained in the column using the relative mass recovery of TiO₂-NPs in the effluents (Table 3). The same theoretical TiO₂-NPs concentration was applied (372 mg kg⁻¹) to the soil columns but the final TiO₂-NPs concentration differed in the 3 exposure scenarios (Table 3). The final highest concentration of TiO₂-NPs was found in the columns submitted to the acute exposure (364 mg kg⁻¹) with 50 mg L⁻¹. In the columns receiving 2 NPs applications at 25 mg L⁻¹, the final TiO₂-NPs concentration was estimated as 350 mg kg⁻¹, which corresponds to a decrease of 6% compared to the acute exposure. After 3 applications at 16.7 mg L⁻¹, only 328 mg kg⁻¹ of

TiO₂-NPs were retained in soil, which corresponds to a decrease of 12% and 6%, compared to the 1 and 2 exposure scenarios, respectively.

Table 3 TiO₂ relative mass recovery in the column effluents for the three applied concentrations in the different exposure scenarios (1, 2 or 3 applications). Final TiO₂-NPs concentration retained in the column is calculated from the relative mass recovery of TiO₂-NPs in the effluents.

TiO ₂ injection concentration	Relative mass recovery of TiO ₂ -NPs:			Final TiO ₂ concentration in soil (mg kg ⁻¹ dry soil)
	1st application	2nd application	3rd application	
50 mg L ⁻¹	2.19%			363.9
25 mg L ⁻¹	4.95%	0.94%		350.1
16.7 mg L ⁻¹	10.66%	0.93%	0.21%	328.1

Effect of acute and chronic contaminations on nitrification activity and ammonia-oxidizer abundance

Two months of incubation consecutive to the last NPs application in soil columns, potential nitrification and AOA and AOB abundances were measured. When considering the whole column (i.e. the 4 soil layers), the acute exposure (i.e. 1 exposure at 50 mg L⁻¹) resulted in a significant increase of the AOA abundance (Figure 1; Table 4) but no NPs effect was observed on AOB abundance and potential nitrification activity (Figure 1; Table 4). After 2 NPs exposures at 25 mg L⁻¹, neither the potential nitrification nor the ammonia-oxidizer abundances were affected by TiO₂-NPs contamination (Figure 1, Table 4). In contrast, potential nitrification, AOA and AOB abundances were significantly decreased after 3 applications of TiO₂-NPs at 16.7 mg L⁻¹ (Figure 1; Table 4).

Table 4 *P*-values from an ANOVA analysis of the effects of TiO₂-NPs, soil depth and their interaction on potential nitrification, AOA and AOB abundances for each exposure scenario. Values in bold are significant at *P* < 0.05.

		Potential nitrification	AOA	AOB
1 exposure	TiO ₂	0.33	<0.001	0.66
	Depth	0.59	0.64	0.19
	TiO ₂ x Depth	0.71	0.74	0.93
2 exposures	TiO ₂	0.38	0.74	0.34
	Depth	0.02	0.59	0.60
	TiO ₂ x Depth	0.38	0.60	0.42
3 exposures	TiO ₂	0.005	0.05	0.01
	Depth	0.003	0.99	0.91
	TiO ₂ x Depth	0.35	0.97	0.87

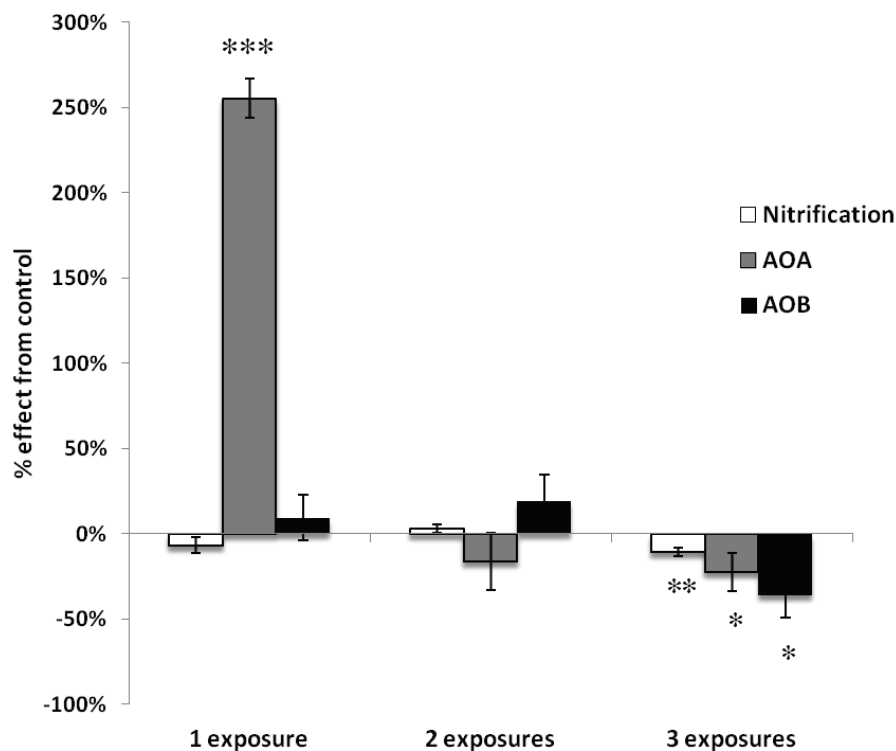


Figure 1 Effect of TiO₂-NPs on potential nitrification activity, AOA and AOB abundance in the whole column (average of the 4 depths). Data are expressed as the percentage change from the control treatment for each exposure. Standard errors are presented (n=6). Significant effects are indicated (***, *P*<0.001; **, *P*<0.01; *, *P*<0.05).

A significant effect of soil depth was only observed on potential nitrification after 2 and 3 exposures ($P=0.02$ and $P=0.003$, respectively; Table 4) and no significant interaction between TiO₂-NPs contamination and soil depth was found (Table 4). After 2 exposures, potential nitrification differed significantly according to soil depth, especially at 2 - 4 cm, but no significant effect of the interaction between TiO₂-NPs and soil depth was observed (Figure 2). After 3 exposures, potential nitrification was significantly more reduced by TiO₂-NPs in the first 2 cm than in 2 – 4 and 4 – 6 cm soil depths (Figure 2).

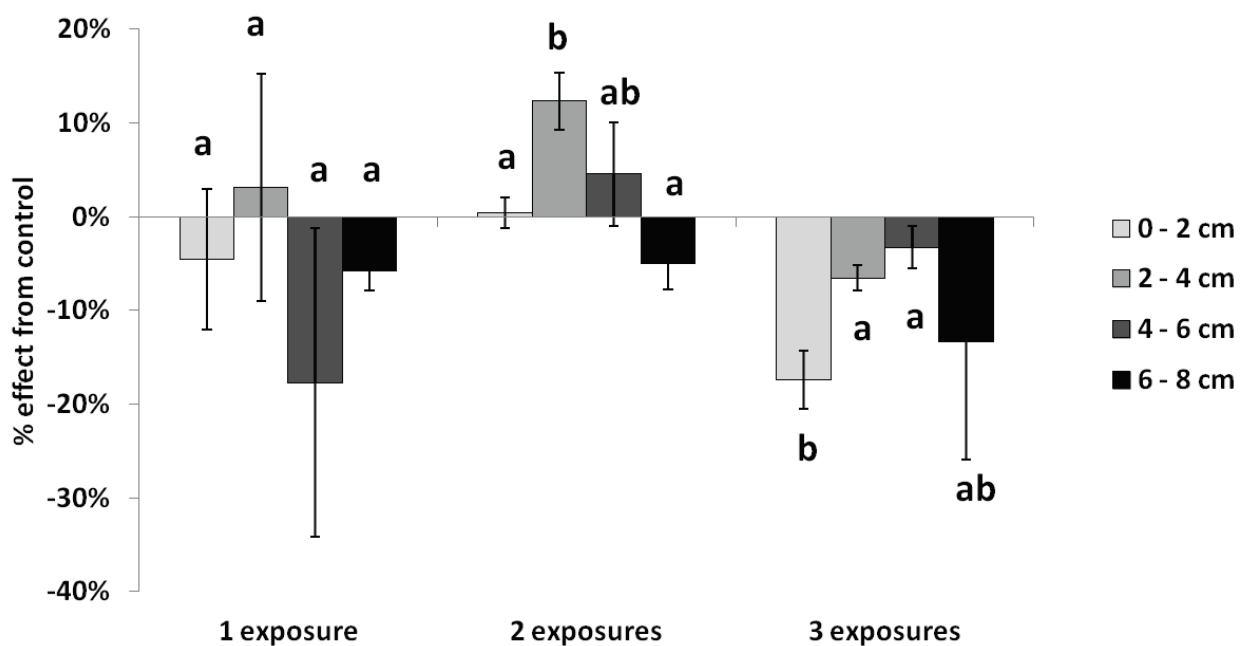


Figure 2 Effect of TiO₂-NPs on potential nitrification activity measured in the four soil layers (depths 0-2 cm; 2-4 cm; 4-6 cm; 6-8 cm) after 1, 2 or 3 soil exposures. Data are expressed as percentage of change from the control treatment for each exposure condition. Standard errors are presented (n=6). For each exposure scenario, bars labeled with the same letter do not differ at $P < 0.05$.

DISCUSSION

Consequences of acute and chronic exposures on TiO₂-NPs transport in soil

Our experimental design enabled the study of the mobility of TiO₂-NPs under acute and chronic exposures for different input concentrations. The results showed that the transport of TiO₂-NPs in soil is influenced by the concentration injected in soil columns for the acute

contaminations. TiO₂ retention increased when higher concentrations were applied. Similar behavior of silica NPs have been observed in sand column by Vitorge *et al.*, (2013) who showed that NPs mobility increased at low concentration regimes. These results can be explained by the increase of TiO₂-NPs aggregation with concentration, which can favor the deposition and straining of these colloids (Solovitch *et al.*, 2010; Cornelis *et al.*, 2014).

Previous studies on TiO₂-NPs mobility in soil reported contrasting results. No transport was observed using 1 g L⁻¹ input concentration of TiO₂-NPs (Nickel *et al.*, 2015), while a significant transport was recorded in soils with low clay content and ionic strength using different concentration of TiO₂-NPs in the different soils studied (Fang *et al.*, 2009). Here we showed that despite the fine texture of the soil (39 % clay and 51 % loam), the transport of TiO₂-NPs can occur after an acute exposure, especially under a low concentration regime which is more likely to be observed in natural soils. In the case of an accidental spill for which a high concentration of NPs is expected, the mobility of TiO₂-NPs will be likely very low, in agreement with a classical blocking mechanism typically observed at high NPs concentrations (Vitorge *et al.*, 2013).

For chronic exposures in the soil columns, a very low mobility of TiO₂-NPs was observed and no concentration effect was identified contrarily to the acute exposure scenario. Although TiO₂-NPs distribution along the soil profiles could not be measured (Nickel *et al.*, 2015), it is likely that most of the TiO₂-NPs were retained in first centimeters of the soil (Navarro *et al.*, 2011). This preferential accumulation at the column inlet could have led to pore occlusion and to stronger filtration of the second and third applications of NPs during chronic exposure. These results observed in small columns of 10 cm length suggest that in soils chronically exposed to TiO₂-NPs, these contaminants will accumulate in the first centimeters of the soil and transport to deeper soil layers and groundwater will probably be limited.

Deleterious effects on nitrification after chronic exposure

Two months after the last TiO₂-NPs application, significant decreases of potential nitrification and ammonia-oxidizer abundance were observed only in soil samples exposed 3 times to NPs. These soil columns were exposed to the lowest TiO₂-NPs concentration (16.7 mg L⁻¹) and the final TiO₂-NPs concentration in soil was slightly lower than in the two other

exposure scenarios. Altogether these results showed for the first time that a soil chronic exposure to NPs could be more toxic for soil microbial community and soil functioning than acute exposures. Furthermore, deleterious effects on nitrification being observed only in soil exposed to 3 applications of low concentrations of NPs suggests that resistance and/or resilience of the microbial community to TiO₂-NPs decreases under short term chronic exposure. Indeed in this experiment, the NPs were applied at intervals of 15 days and the effects on nitrification were assessed after a 2 months incubation period. It is well known that a microbial community can develop a tolerance to various stressors over time through modifications of community composition resulting in the selection of tolerant populations and the dispersal of specific resistance via mobile genetic elements (Bissett *et al.*, 2013; Griffiths and Philippot, 2013; Allison and Martiny, 2008). It is likely that chronic exposures occurring on the long term will have less consequences on nitrification because of potential microbial community adaptation over time, as it was observed by Mertens *et al.* (2009) after 2 years of exposure to zinc pollution. To get a better knowledge of NPs impact on soil functioning after chronic contamination, it would be interesting to study the NPs effects on microbial activity, abundance and diversity after each exposure and before each new exposure, in order to monitor the microbial community resistance and to assess resilience, respectively. This approach may also lead to new insights on sensitive and tolerant microbial species among nitrifiers and thus on TiO₂-NPs toxicity mechanisms in soil.

After 3 NPs applications, it is likely that most of TiO₂-NPs were retained on the top of the column. Although a TiO₂ concentration gradient may be present along the soil column profile, no effect of depth was noticed on ammonia-oxidizer abundance that were decreased similarly at the 4 soil depths. Nitrification was significantly more decreased in the first 2 cm and the last 2 cm than in the 2 – 4 and 4 – 6 cm soil layers. With common pollutants it is usually expected that high concentrations have the most deleterious effects. In the case of NPs it has been demonstrated that TiO₂-NPs concentrations induce nonlinear dose-responses on potential nitrification, as well as on AOA and AOB abundance (Article 5). Very low (0.05 mg kg⁻¹) and high concentrations (100 – 500 mg kg⁻¹) can result in similar decreases, which could likely explain the absence of clear effect of soil depth on the response of nitrifiers to TiO₂ applications. These results suggest that despite the low TiO₂ mobility in soil, microbial community can be affected by TiO₂-NPs at the depths of

preferential NPs accumulation (top soil) but also at depths reached by NPs at lower concentration.

CONCLUSIONS

The multidisciplinary approach developed in this study enabled to compare the impact of realistic acute and chronic TiO₂-NPs contaminations in soil columns. The transport of TiO₂-NPs in the studied silty-clay soil was very low for both acute and chronic exposures and varied with NPs injected concentration. In these conditions, the transport of TiO₂-NPs to deeper soil layers or groundwater is likely limited.

Although the chronic exposure resulted in a lower final TiO₂-NPs concentration, it was more toxic to soil microbial community and soil functioning compared to the acute exposure. Chronic exposure in short and long term experiment representing realistic scenarios of soil exposure to NPs are needed to study the resistance and resilience of soil microbial communities to these emerging contaminants and their subsequent consequences on soil fertility.

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5. Conclusions du chapitre

Dans ce chapitre, nous avons montré que la nitrification était un processus microbien sensible à une pollution aux TiO₂-NPs, même pour des concentrations très faibles (0,05 mg kg⁻¹) et dans des contextes d'exposition aiguë ou chronique. Les diminutions de l'activité nitrifiante ont été expliquées par une forte sensibilité des AOA à ce contaminant et à leur rôle clé dans l'oxydation de l'ammonium dans le sol étudié. Dans ce sol, les AOA étaient principalement représentées par un OTU affilié au genre *Nitrososphaera* mettant en évidence une faible diversité et redondance fonctionnelle au sein de ce groupe. Toutefois, il existe une redondance fonctionnelle entre les AOA et les AOB, ainsi qu'au sein des NOB, qui semble limiter les effets délétères du TiO₂ sur l'activité nitrifiante globale.

Les effets négatifs observés sur l'activité dénitrifiante sont des effets indirects liés à la diminution de l'activité nitrifiante en amont. Le constat réalisé pour d'autres polluants indiquant une plus grande résistance des microorganismes dénitrifiants (comparés aux nitrifiants) semble être vérifiée également dans le cas des TiO₂-NPs.

Nos travaux sur l'écotoxicologie des TiO₂-NPs, se basant sur la nitrification comme indicateur, ont permis de montrer qu'il n'existait pas de relation dose-réponse linéaire contrairement à ce qui peut être observé dans le cas de polluants solubles, tels que les métaux lourds ou les pesticides (Giller et al. 1998 ; Guo et al. 2011). Ce fait est expliqué par l'influence de la concentration utilisée sur les caractéristiques physico-chimiques des NPs, en particulier au niveau de leur agrégation et de leur potentiel oxydant. Ces paramètres sont probablement impliqués directement dans la biodisponibilité et toxicité des TiO₂-NPs qui reposent notamment sur leur capacité à générer des ROS et à s'adsorber à la surface des membranes microbiennes. Le fait que les propriétés des NPs varient avec la concentration utilisée incite à réaliser une caractérisation de ces polluants à toutes les doses testées, contrairement à ce qui est fait habituellement. De plus, il ne semble pas pertinent, dans le cas des NPs, de baser l'évaluation des risques dans les sols sur des relations dose-réponse à partir desquelles sont calculées des EC50, LC50 ou d'autres concentrations définissant des seuils de toxicité.

Nous avons pu voir également que les phénomènes liés à la concentration des NPs ont des conséquences sur leur transport dans les sols, en conséquence d'une modification de

l'agrégation et de leur réactivité vis-à-vis des surfaces réactives du sol. Les NPs les moins agrégées étant plus mobile dans la porosité du sol.

Enfin, nous avons montré qu'une exposition chronique des sols aux NPs était plus toxique pour les microorganismes nitrifiants qu'une exposition aiguë lors d'une contamination en colonne. Ces premiers résultats suggèrent que la résistance et/ou la résilience des communautés microbiennes sont affectées par le nombre d'expositions réalisées et il serait intéressant d'étudier les modifications de structure de communauté, d'abondance et d'activité au cours de chacune de ces expositions.

Nous avons pu aussi constater que le mode d'exposition utilisé pour contaminer le sol avec les TiO₂-NPs a son importance. En effet, une contamination aiguë à 364 mg kg⁻¹ effectuée à la surface du sol en colonne n'a pas eu d'effet négatif sur la nitrification (Article 6), alors qu'en microcosmes du même sol lors d'une contamination aiguë « homogène » à 100 et 500 mg kg⁻¹, l'activité était réduite d'environ 25% (Article 5). Ainsi lors de contaminations aiguës dans des conditions plus réalistes d'exposition, nos résultats indiquent que les TiO₂-NPs à forte concentration n'auraient plus d'effets délétères sur le fonctionnement microbien du sol étudié.

Chapitre 4 : Conclusions générales et perspectives

L'objectif de ces travaux était d'évaluer les risques associés à la dissémination de NPs d'oxydes métalliques dans les sols, au travers d'une approche pluridisciplinaire se basant sur l'écotoxicologie, l'écologie fonctionnelle et l'écodynamique de ces contaminants. Les indicateurs écologiques choisis ont permis de rendre compte des effets à l'échelle de communautés naturelles d'organismes réalisant des processus clés impliqués dans des services écosystémiques rendus par les sols. Différentes questions ont été adressées au début de la thèse et dans cette partie conclusive, nous reviendrons sur chacune d'entre elle afin de tirer des conclusions générales et de proposer des perspectives de recherche.

1. Influence des propriétés du sol sur le devenir et l'écotoxicité des NPs d'oxydes métalliques

- **Quelle est l'influence des propriétés du sol, notamment la texture et la teneur en matière organique, sur la mobilité des NPs d'oxydes métalliques ?**

Globalement, la mobilité du TiO_2 et CuO observée dans les 6 sols est très faible car au minimum 85% des NPs injectées sont retenues dans le sol. Comme attendu, nous avons constaté que les propriétés du sol influençaient les caractéristiques physico-chimiques et par conséquent le transport des NPs d'oxydes métalliques. Cependant, contrairement à ce qui peut être observé avec des polluants solubles, nous n'avons pas pu dégager une typologie de sol dans laquelle la mobilité des NPs serait favorisée, en raison de l'absence de relation claire entre la texture du sol et la mobilité des NPs. En se basant sur la littérature, on pouvait s'attendre à un transport plus important dans les sols sableux, mais que l'on considère les NPs de TiO_2 ou de CuO , cela n'a pas été vérifié, ce qui en soit est un résultat très intéressant. La teneur en argile du sol ne semble donc pas être un paramètre clé du transport des NPs dans les sols étudiés. En revanche, la teneur en MO apparaît comme un paramètre d'intérêt, en particulier le carbone organique dissous qui influence l'agrégation, les charges de surface et au final la dispersion et la mobilité des CuO -NPs dans les sols. Le transport du CuO était accru dans des sols ayant des concentrations en carbone organique dissout élevées indépendamment de la texture (sablo-limoneux et limono-argileux).

Ces résultats montrent la nécessité d'étudier plus spécifiquement le rôle de la MO dissoute, en termes de qualité et de quantité, sur les caractéristiques des NPs et leur mobilité dans les

Chapitre 4 : Conclusions générales et perspectives

sols. Des analyses sont en cours pour caractériser la MO des sols et solutions de sol utilisées, par spectroscopie en proche infrarouge (collaboration Lauric Cecillon-IRSTEA Grenoble).

L'étude du TiO_2 dans les sols est très complexe du fait de la forte teneur en titane des sols et des solutions de sols. De ce fait, les courbes de percée n'étaient pas exploitables pour la comparaison entre sols et pour la modélisation. Toutefois, nous avons pu mettre en évidence que la concentration en TiO_2 -NPs utilisée a des conséquences pour leur transport dans les sols, certainement via une modification de leur agrégation, les TiO_2 -NPs les moins agrégées étant plus mobiles dans la porosité du sol.

Pour améliorer les connaissances du transport du TiO_2 dans les sols, l'utilisation de nanotraceurs contenant un marqueur permettant leur détection (e.g. fluorescents, radioactifs, magnétiques ; Vitorge, 2010 ; Hildebrand et Franke, 2012) serait une solution à considérer.

L'étude de l'influence de la concentration en NPs sur leur transport dans différents sols naturels sera également à poursuivre dans de futures études. Cet aspect est d'autant plus important qu'en fonction du mode d'exposition des sols, par l'intermédiaire d'épandage de boues, l'utilisation de NPs dans des produits sanitaires ou fertilisants ou encore lors d'un déversement accidentel, la concentration en NPs sera très différente, induisant sans doute des comportements différents.

A l'heure actuelle, il reste de nombreuses zones d'ombres concernant le devenir des NPs dans les sols. Toutefois, il est clair que les NPs sont très peu mobiles dans les sols et sont donc susceptibles de persister pendant de longues durées dans cet écosystème. Ce contexte implique la prise en considération de leur vieillissement/transformation au cours du temps, mais aussi de leur remobilisation dans les sols sur des échelles de temps beaucoup plus longues que celles généralement étudiées, ce qui ouvre donc également des perspectives intéressantes.

- **Quelle est l'influence des propriétés du sol, en particulier de la texture et de la teneur en matière organique, sur la toxicité des NPs d'oxydes métalliques sur les communautés microbiennes du sol?**

Les TiO₂-NPs n'ont pas induit d'effets toxiques sur les communautés microbiennes dans la majorité des sols. Des effets délétères sur le fonctionnement microbien n'ont été observés que dans un sol sur les six étudiés. Il s'agit du sol limono-argileux (forte MO) qui présentait le pH et la teneur en MO les plus élevés. Ces résultats peuvent paraître surprenants car il a été mis en évidence dans la littérature que les communautés microbiennes de sols ayant des teneurs élevées en argile et MO étaient plus résistants et résilients aux perturbations, en particulier aux métaux lourds (Giller *et al.*, 1998 ; Debeljak *et al.*, 2009 Griffiths et Philippot 2013). Dans cette étude, le résultat inverse a donc été observé, mettant en évidence que les connaissances issues de l'étude des métaux lourds ne sont pas transposables simplement aux NPs métalliques.

Dans la solution du sol limono-argileux, les TiO₂-NPs avaient des caractéristiques physico-chimiques singulières comparées aux autres sols, pouvant probablement expliquer leur plus forte toxicité. En effet, en raison des caractéristiques de la solution du sol (force ionique, pH, carbone organique dissous, calcium), les TiO₂-NPs étaient moins stables en suspension. Il est vraisemblable que dans ces conditions, l'hétéroagrégation soit favorisée ce qui peut conduire à l'adsorption des NPs sur les membranes microbiennes. Ces résultats montrent que les interactions complexes entre NPs et composés du sol rendent difficiles la mise en évidence des facteurs favorisant leur toxicité. Des études complémentaires seraient nécessaires pour déterminer l'influence de la stabilité des TiO₂-NPs sur leur biodisponibilité et donc leur toxicité dans les sols. Une démarche réductionniste consistant à évaluer la toxicité des NPs dans des conditions simplifiées en solution de sol serait sans doute adaptée pour tester cette hypothèse.

Il serait également intéressant de tester les effets des TiO₂-NPs sur un plus grand nombre de sols afin d'avoir une puissance statistique importante pour évaluer l'influence des propriétés du sol sur leur toxicité. Dans le panel de sols choisi, il faudrait sélectionner des sols qui présentent des caractéristiques très contrastées, mais également des sols aux propriétés similaires, afin de pouvoir différencier les effets liés aux paramètres physico-chimiques et ceux liés à la sensibilité de certaines communautés microbiennes vis-à-vis de ce polluant.

Cela permettrait en particulier de déterminer si le pH et la teneur en MO sont réellement des paramètres dominants influençant la biodisponibilité et toxicité du TiO₂.

Les effets observés sur les indicateurs microbiens reposent sur l'influence des propriétés du sol sur la toxicité du polluant mais également sur la résistance de la communauté microbienne exposée. Les effets délétères du TiO₂ dans le sol limono-argileux pourraient donc être une conséquence de la plus grande sensibilité de la communauté microbienne de ce sol comparée à celle des autres sols. Pour vérifier cela, on pourrait envisager d'extraire des communautés microbiennes à partir de différents sols puis de les exposer aux TiO₂-NPs dans un même environnement afin de comparer leur résistance.

En conclusion, les propriétés du sol influencent le devenir et la toxicité des NPs, mais à ce stade des recherches, il ne paraît pas possible d'identifier une typologie de sols présentant une vulnérabilité particulière aux NPs. Au vu de la complexité des interactions et des transformations subies par les NPs dans les sols et des nombreux facteurs qui conditionnent la réponse des communautés microbiennes, il serait même légitime de s'interroger sur le réalisme de cet objectif. Bien que les effets toxiques des TiO₂-NPs aient été mis en évidence dans un seul sol sur les six étudiés, le principe de précaution inciterait à ce qu'une réglementation en faveur de la protection des sols soit proposée pour ce type de composés, notamment car leur comportement et toxicité dans les sols est difficilement prévisible à l'heure actuelle.

2. Impact des TiO₂-NPs sur le cycle de l'N

- **Quel est l'effet de ces NPs sur les groupes fonctionnels microbiens nitrifiants et dénitrifiants impliqués dans le cycle de l'azote dont les niveaux de redondance fonctionnelle sont différents?**

Dans le sol limono-argileux, des effets délétères sur les activités microbiennes du cycle de l'N ont été observés. Comme nous en avons fait l'hypothèse, le processus microbien le plus sensible aux TiO₂-NPs est la nitrification du fait de la faible redondance fonctionnelle des groupes fonctionnels microbiens impliqués. En conséquence de la diminution de l'activité nitrifiante, la dénitrification était réduite mais dans une moindre mesure et aucune

Chapitre 4 : Conclusions générales et perspectives

modification de l'abondance des dénitrifiants n'a été observée. Nos résultats suggèrent que les TiO₂-NPs peuvent avoir des conséquences globales sur le cycle de l'N, en particulier sur la disponibilité en NH₄⁺ et NO₃⁻ qui sont 2 sources clés pour la nutrition végétale.

Dans le futur, un changement d'échelle d'étude sera nécessaire afin (i) d'évaluer les conséquences indirectes des effets sur le cycle de l'N sur la croissance végétale et (ii) d'étudier comment les plantes modulent les effets des NPs sur les communautés microbiennes, en particulier via l'exsudation racinaire car ces composés organiques pourraient modifier la biodisponibilité et la toxicité des NPs (Martineau *et al.*, 2014). Un projet permettant d'aborder ces questions en utilisant des mésocosmes plantés a été déposé à la fondation Rovaltain (RT2E-2015) afin de poursuivre les investigations initiées dans ce travail.

Dans le sol étudié, les AOA semblent être un groupe microbien « clé de voute » pour le fonctionnement du cycle de l'azote. Paradoxalement, ce groupe fonctionnel se caractérise par une très faible diversité avec la dominance du genre *Nitrososphaera* et donc une très faible redondance fonctionnelle pour la réalisation de l'oxydation de l'ammonium. Suite à ce constat, on peut se demander si les sols dans lesquels la nitrification repose principalement sur l'activité des AOA sont moins résistants et résilients aux perturbations, et aux polluants en particulier. En effet, la faible diversité des archées totales et AOA a été mise en évidence dans différentes études (Bates *et al.*, 2010 ; Pester *et al.*, 2012) et il a été montré que la restauration de l'activité nitrifiante dans un sol contaminé au Zn était assuré par les AOB et pas les AOA (Mertens *et al.*, 2009). Il faut toutefois noter que les outils moléculaires pour étudier les archées, notamment les amorces de PCR, sont en constante amélioration et évaluation et qu'il est possible que les résultats obtenus à l'heure actuelle ne reflète pas la diversité complète de cette communauté (Pester *et al.*, 2012). Par ailleurs, le faible nombre de souches isolées du sol (Lehtovirta-Morley *et al.*, 2011 ; Tournia *et al.*, 2011) limite la connaissance sur la physiologie et l'écologie de ces procaryotes (Prosser et Nicol, 2012). La différenciation de niches entre AOA et AOB n'est pas encore claire et il est vraisemblable qu'elle repose sur une combinaison de facteurs environnementaux, tels que le pH, la MO et la concentration en ammonium (Erguder *et al.*, 2010 ; Prosser et Nicol, 2012 ; Zhalnina *et al.*, 2012). Au vu des différents niveaux de diversité présents dans ces 2 groupes, on pourrait se demander si le degré de perturbation d'un sol (e.g. pollutions, pratiques agricoles...) ne

Chapitre 4 : Conclusions générales et perspectives

serait pas un paramètre supplémentaire à prendre en considération pour comprendre l'abondance relative et le rôle fonctionnel des AOA et AOB dans les sols (Vico Oton *et al.*, 2015). Pour étayer cette hypothèse, il conviendrait dans un premier temps d'identifier des sols non-perturbés dans lesquels les AOA ont un rôle fonctionnel majeur, puis d'exposer ces sols à différents types de perturbations pour déterminer si cela conduit à ce que les AOB deviennent dominants.

L'approche statistique par path analysis a contribué à révéler les effets directs et indirects des NPs, ainsi que les liens fonctionnels entre les différents acteurs de la nitrification et dénitrification. Cette approche offre une vision intégrée de l'impact d'un polluant sur les communautés microbiennes et leur fonctionnement. Elle a permis de mettre en évidence les effets négatifs en cascades dus à la sensibilité des AOA du genre *Nitrososphaera*, ainsi que les altérations de la nitrification expliquées par des modifications de couplage entre les groupes fonctionnels impliqués dans la première et la seconde étape du processus. Ces résultats confirment l'intérêt en écotoxicologie d'une approche systémique intégrant la structure des communautés, les activités, les fluctuations temporelles, les interactions et les feedbacks (Bisset *et al.*, 2013).

Nous n'avons pas intégré de facteurs abiotiques (T°C, humidité, pH...) dans les path analysis en raison de leur stabilité dans les microcosmes. Cependant, dans des expérimentations en mésocosmes, il sera intéressant d'intégrer plus de variables abiotiques et biotiques d'intérêt dans ce type d'approche statistique en vue d'élucider la complexité du système. Certains aspects non étudiés dans la thèse pourront être abordés, tels que les effets indirects des NPs sur les propriétés du sol via la modification de la chimie de la solution du sol ou la remobilisation de polluants (Ben Moshe *et al.*, 2013 ; Priester *et al.*, 2013 ; McShane *et al.*, 2014).

Les groupes fonctionnels nitrifiants sont donc des indicateurs microbiens sensibles à la présence des TiO₂-NPs dans les sols qui peuvent entraîner des conséquences plus larges sur la dénitrification. Ces résultats questionnent quant à l'impact global des TiO₂-NPs sur le fonctionnement des écosystèmes, car dans cette étude nous nous sommes intéressés uniquement aux processus de minéralisation du carbone, de nitrification et de dénitrification. Nos résultats suggèrent que le service écosystémique de maintenance de la

fertilité par les communautés microbiennes pourrait être altéré, mais des expérimentations à plus grande échelle doivent encore être menées pour valider ces conclusions.

- **Les faibles concentrations ont-elles des effets négatifs sur le fonctionnement microbiologique du sol ? Existe-t-il une relation dose-réponse associée à l'impact des NPs sur les communautés microbiennes du sol ?**

Des effets négatifs de très faibles concentrations en TiO₂-NPs (0.05 mg kg⁻¹) ont été mis en évidence sur l'activité nitrifiante dans le sol limono-argileux. Cependant, en étudiant l'effet de ce contaminant sur différents indicateurs microbiens, aucune relation dose-réponse « classique » n'a été observée. Ceci semble être expliqué par l'influence de la concentration utilisée sur les caractéristiques physico-chimiques des NPs, en particulier au niveau de leur agrégation et de leur potentiel oxydant. Ainsi, de façon contre-intuitive, une forte concentration en NPs pourrait être moins biodisponible et moins toxique qu'une faible concentration dans les sols. Ce résultat remet donc en question les approches d'évaluation des risques des NPs dans les sols par la définition de concentrations d'effets et de concentrations seuils (EC₅₀, LOEC...) qui se basent sur les relations dose-réponses. Pour le moment, les expérimentations ayant été effectuées dans un seul sol, il est nécessaire de renouveler l'étude des relations dose-réponse dans des sols présentant des propriétés contrastées pour confirmer ce résultat. Dans ce contexte, l'étude de la toxicité de gammes très larges de concentrations sera à vérifier en intégrant des concentrations faibles (1 µg à 1 mg kg⁻¹) qui sont encore peu testées, alors que ces doses de NPs sont prédites dans les sols (Sun *et al.*, 2014).

Sur différents indicateurs, nous avons observé des effets négatifs qui restent stables ou qui augmentent au cours du temps. Les expériences menées durant 90 jours d'exposition ont rarement mis en évidence une résilience des effets. La majorité de la littérature existante a évalué les effets à court terme des NPs (Simonin et Richaume, 2015), de ce fait nous nous attendions plutôt à des effets sur le court terme, puis à une résilience suite au vieillissement des NPs dans les sols et la sélection de populations tolérantes. Les taux de croissance des micro-organismes dans les sols sont faibles (Demoling *et al.*, 2007), comparés à ceux mesurés *in vitro*, particulièrement dans le cas des micro-organismes autotrophes impliqués dans la nitrification. Ce fait pourrait expliquer pourquoi certains effets sur l'abondance ou la

diversité ne sont détectés qu'après 90 jours. De plus, l'ADN extrait peut provenir de cellules mortes en raison de sa persistance possible pendant plusieurs semaines dans le sol (Niemeyer et Gessler, 2002), ce qui a pu empêcher la détection d'effets à court terme sur l'abondance et la diversité microbienne. Des études sont encore nécessaires pour évaluer les effets des NPs dans les sols sur le long terme (plusieurs mois ou années) suite à leur transformation/vieillessement et établir si une résilience des effets est possible et dans quel délai.

3. Effet d'une pollution chronique

- **Le transport et la toxicité des NPs dans les sols sont-ils modifiés lors d'apports chroniques et aigus ?**

Les contaminations chroniques conduites en colonnes de sols ont montré que les NPs sont mobiles majoritairement lors de la première contamination et qu'elles sont par la suite retenues en quasi-totalité dans le sol. De ce fait pour une même concentration injectée, nous avons mis en évidence que la concentration en TiO_2 -NPs retenue dans le sol est plus grande dans le cas de contamination aiguë que lors d'une contamination chronique. Toutefois, les effets délétères sur la nitrification ont été observés uniquement dans le cas de trois expositions successives aux NPs résultant en une concentration en TiO_2 -NPs plus faible dans le sol que dans les autres scénarios d'exposition. Ces observations fournissent des éléments nouveaux qui méritent d'être pris en compte en matière de risques associés aux NPs dans les sols. Il s'agit en effet d'une toute première étude mettant en évidence que les expositions chroniques sont plus toxiques que les expositions aiguës pour les communautés microbiennes. Dans un contexte où les épandages de boues, l'irrigation ou l'utilisation de nanofertilisants engendrent vraisemblablement une exposition chronique des sols agricoles, un tel constat mérite une attention particulière.

Il est donc nécessaire de poursuivre ce travail en considérant d'autres sols, une gamme de concentrations testées plus large, des durées d'expositions plus longues, d'autres indicateurs et un nombre de contaminations plus important afin de valider la généralité de ces résultats. Un transfert d'échelle expérimentale, permettant de considérer une distance de parcours plus réaliste que les 10 cm utilisés, sera également une étape indispensable.

Chapitre 4 : Conclusions générales et perspectives

Le mode de contamination des sols est également un élément clé à considérer dans le futur. En effet, une contamination du sol « homogène » dans les microcosmes et une contamination apportée en surface dans une colonne ont des conséquences différentes sur la nitrification. La contamination aiguë en colonne de sol n'a eu aucun effet sur la nitrification contrairement à ce qui a été observé en microcosmes, alors que le sol utilisé pour ces expériences avait été prélevé au même moment. De ce fait, il est possible que les expositions réalisées en microcosmes ne soient pas suffisamment représentatives de la toxicité des NPs dans cet environnement. La très forte réactivité des NPs nécessite d'adapter les protocoles d'analyse de l'écotoxicité des NPs dans les sols. Nos résultats vont dans ce sens, ce qui incite donc à mettre en œuvre des systèmes expérimentaux s'approchant au mieux de conditions réalistes d'expositions afin d'améliorer la fiabilité de l'évaluation des risques associés aux NPs dans le futur.

L'évaluation des risques associés aux NPs nécessiterait à moyen terme la réalisation d'études de terrain sur des sites de pollution aux NPs (usine, déversement accidentel...), afin d'étudier directement l'effet d'une contamination aiguë ou chronique sur le long terme dans des conditions naturelles. Ces suivis de terrain permettraient de valider la pertinence des différents indicateurs microbiens utilisés dans la thèse et plus généralement dans la littérature. Ils se révéleraient également très utiles pour caractériser les NPs et leur transport *in situ*. De plus, ces sols contaminés pourraient être collectés pour être à nouveau exposés au laboratoire afin d'étudier la résilience et la résistance des communautés microbiennes, et identifier des communautés adaptées à ces perturbations.

La problématique des NPs se pose également dans un cadre plus large de multipollutions qui constituent un des défis scientifiques majeurs actuels. Il conviendrait donc d'envisager leur impact en association avec d'autres polluants comme les métaux lourds, les pesticides ou les antibiotiques, dans la mesure où ces derniers partagent les mêmes voies d'entrées dans le sol, par exemple via les boues de station d'épuration. Les interactions entre les NPs et d'autres polluants font déjà l'objet d'applications, par exemple en remédiation ou retraitement des eaux (Fang *et al.*, 2011 ; Hua *et al.*, 2012). Il est donc probable que la toxicité de tels cocktails ne résulte pas seulement d'effets additifs, mais plus vraisemblablement d'effets synergiques ou antagonistes associés à des interactions complexes.

4. Une approche « systems biology » au service de l'écotoxicologie

Durant les 20 dernières années, les approches de type « systems biology » ont émergé comme une discipline scientifique holistique qui vise à comprendre comment les interactions s'opèrent à différents niveaux d'organisation biologique (e.g. de l'ADN/ARN/protéine aux cellules, organismes, jusqu'à la structure des communautés et le fonctionnement) et les paramètres environnementaux (e.g. nutriments, humidité, jusqu'aux conditions climatiques locales et globales) donnent naissance aux processus biologiques et écologiques (Bisset *et al.*, 2013 ; Raes et Bork, 2008). Ces travaux de thèse s'inscrivent en partie dans cette démarche mais en l'appliquant à l'écotoxicologie. Cette approche apporte un éclairage nouveau sur l'écotoxicologie en permettant de rendre compte des effets de perturbations à l'échelle du fonctionnement des communautés, dans le but final de déterminer les impacts sur des services écosystémiques tels que la maintenance de la fertilité du sol et de la biodiversité (Millenium ecosystem assessment, 2005). Ainsi, l'écotoxicologie ne se cantonne plus seulement à rendre compte d'effets sur des variables isolées et à définir des concentrations seuils et des relations dose-réponse, mais tend vers une meilleure compréhension de l'influence de perturbations sur le fonctionnement des écosystèmes en intégrant différentes échelles d'études (cellule, population, communauté) et les interactions biotiques et abiotiques.

L'intégration d'une démarche de type « systems biology » permet ainsi d'orienter l'écotoxicologie vers de l'écologie du stress ce qui répond aux nombreux appels à inclure plus d' « éco » en écotoxicologie (Calow, 1996 ; Van Straalen, 2003) alors que dans les faits, encore actuellement, l'écotoxicologie est plus souvent synonyme de toxicologie environnementale (Van Straalen, 2003).

Dans cette thèse, la démarche intégrative que nous avons adoptée a permis d'apporter une contribution pluridisciplinaire originale pour la compréhension de l'impact des NPs d'oxydes métalliques. Toutefois, la route est encore longue avant que le devenir et la toxicité des nanomatériaux dans les sols soient bien compris et qu'une réglementation adaptée à cet environnement soit proposée.

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