

*Soil pH effects on the interactions  
between dissolved zinc, non-nano- and  
nano-ZnO with soil bacterial communities*

Article

Accepted Version

Read, D. S., Matzke, M., Gweon, H. S., Newbold, L., Heggelund, L., Ortiz, M. D., Lahive, E., Spurgeon, D. and Svendsen, C. (2016) Soil pH effects on the interactions between dissolved zinc, non-nano- and nano-ZnO with soil bacterial communities. *Environmental Science and Pollution Research*, 23 (5). pp. 4120-4128. ISSN 1614-7499 doi: <https://doi.org/10.1007/s11356-015-4538-z> Available at <https://centaur.reading.ac.uk/75777/>

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To link to this article DOI: <http://dx.doi.org/10.1007/s11356-015-4538-z>

Publisher: Springer

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1 Soil pH effects on the interactions between dissolved zinc, non-nano  
2 and nano ZnO with soil bacterial communities

3

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16

17

1 *Abstract*

2 Zinc oxide nanoparticles (ZnO NPs) are used in an array of products and processes, ranging from  
3 personal care products to antifouling paints, textiles, food additives, antibacterial agents and  
4 environmental remediation processes. Soils are an environment likely to be exposed to manmade  
5 nanoparticles due to the practice of applying sewage sludge as a fertiliser or as an organic soil  
6 improver. However, understanding on the interactions between soil properties, nanoparticles and the  
7 organisms that live within soil is lacking, especially with regards to soil bacterial communities. We  
8 studied the effects of nanoparticulate, non-nanoparticulate and ionic zinc (in the form of zinc chloride)  
9 on the composition of bacterial communities in soil with a modified pH range (from pH 4.5 to pH 7.2).  
10 We observed strong pH dependent effects on the interaction between bacterial communities and all  
11 forms of zinc, with the largest changes in bacterial community composition occurring in soils with low  
12 and medium pH levels (pH 4.8 and 5.9). The high pH soil (pH 7.2) was less susceptible to the effects of  
13 zinc exposure. At the highest doses of zinc (2500 mg/kg dw soil) both nano and non-nano particulate  
14 zinc applications elicited a similar response in the soil bacterial community, and this differed  
15 significantly to the ionic zinc salt treatment. The results highlight the importance of considering soil  
16 pH in nanotoxicology studies, although further work is needed to determine the exact mechanisms  
17 controlling the toxicity and fate and interactions of nanoparticles with soil microbial communities.

## 1 *Introduction*

2 Zinc oxide nanoparticles (ZnO NPs) are amongst those with the highest production volume and  
3 therefore a high environmental release potential. ZnO NP applications range from personal care  
4 products (e.g. sun screen, toothpaste) to antifouling paints, textiles, food additives, antibacterial  
5 agents and environmental remediation processes (Lopes et al. 2014). As a result, sewage treatment  
6 works have been identified as having a high potential burden, with sludge containing up to 17.1 mg/kg  
7 dry weight (dw) followed by sewage sludge treated soils with 3.25 mg/kg dw (compared to average  
8 soil concentrations predicted to be around 0.093 mg/kg dw) (Gottschalk et al. 2009). However these  
9 values are likely to further increase with the increasing use of ZnO NPs in consumer products and  
10 processes.

11 Single species studies have highlighted varying sensitivity of bacteria to zinc nanoparticles (Baek &An  
12 2011, Brayner et al. 2006, Dimkpa et al. 2011, Hernandez-Sierra et al. 2008, Jiang et al. 2009, Jones et  
13 al. 2008, Mortimer et al. 2008, Negi et al. 2012). These studies have been used to provide an insight  
14 into the mechanisms of nanoparticle toxicity, and have indicated the role of reactive oxygen species  
15 (ROS) (Choi &Hu 2008, Kumar et al. 2011b, Lu et al. 2012), DNA damage (Kumar et al. 2011b, a),  
16 oxidation of proteins, interruption of energy transduction and photo catalytic oxidation (Gou et al.  
17 2010, Klaine et al. 2008). Although single species studies are highly informative in terms of modes of  
18 toxic action, they lack the complexity of natural systems comprising hundreds to thousands of  
19 interacting species (Roesch et al. 2007). Soil microorganisms form the base of soil ecosystems as they  
20 are the key players of many soil functions such as biogeochemical cycling, plant productivity and  
21 climate regulation. Estimated numbers are in the range of  $10^9$  cells/g soil with a diversity of  $10^4$  species  
22 (Griffiths and Philppot, 2013). Including environmentally more realistic systems into the hazard  
23 assessment of nanoparticles will consequently lead to a more complete environmental risk  
24 assessment of engineered nanomaterials.

25 Previous work has focused on wastewater systems, identified as potential accumulators of engineered  
26 nanoparticles, has highlighted effects on ecological functioning including methane production,  
27 nitrogen and phosphorous removal, the production of extracellular polymeric substances (EPS) and  
28 protease activity (Choi &Hu 2008, Mu &Chen 2011, Mu et al. 2011, Mu et al. 2012, Musee et al. 2011,  
29 Zheng et al. 2011). The toxicity of nanoparticles in natural systems has been shown not only to be  
30 species dependent, but reliant on various factors to do with the properties of the nanoparticles  
31 themselves (e.g. nanoparticle composition, size, surface area and shape), but also their interaction  
32 with the environment (pH and availability of natural organic matter) (Musee et al. 2011). Studies  
33 investigating the impact of nanomaterials on soil microbial communities are still relatively scarce due

1 to the complex nature of the topic. However, they do include a diverse range of nanomaterials such  
2 as silver, cerium oxide, iron oxide, copper, carbon based nanomaterials, silica, zinc and titanium (Ge  
3 et al. 2011, Kumar et al. 2011b, a) in a wide variety of measured endpoints including; diversity and  
4 modification of the community composition, substrate induced respiration, DNA quantity or impact  
5 on enzyme activities. The response of soil microbial communities to the addition of zinc oxide (ZnO)  
6 nanoparticles has been shown to be highly varied and dependent upon the properties of the systems  
7 and the form of zinc that is applied. These include reductions in both microbial biomass and diversity  
8 (Ge et al. 2011), growth inhibition (Rousk et al. 2012) and species-specific responses to zinc exposure  
9 (Collins et al. 2012, Ge et al. 2012).

10 While some variation in biotic responses may arise from the varying composition of microbial  
11 communities being tested, it is becoming apparent that variation also arises from differences in both  
12 the nanoparticles being used and how different exposure media affects the speciation and  
13 characteristics of NPs once in the soil environment. As well as the characteristics of NPs (e.g., size,  
14 shape, surface charge), other soil properties including organic matter (Coutris et al. 2012, Waalewijn-  
15 Kool et al. 2014), clay contents (Fang et al. 2009), soil moisture (Ge et al. 2013) and the presence of  
16 plants (Ge et al. 2014) have been shown to affect physical and chemical processes, resulting in differing  
17 rates of NP dissolution, agglomeration, and aggregation (Tourinho et al. 2012). However, soil pH  
18 appears to be one of the key factors affecting the behaviour of metal NPs in soil and recent work has  
19 shown the role this plays in NP dissolution, mobility and bioavailability to soil organisms (Heggelund  
20 et al. 2014, Waalewijn-Kool et al. 2013).

21 The aim of this study was to elucidate the role that soil pH plays on mediating the effect of zinc on soil  
22 bacterial communities, and; whether the form in which zinc is dosed (nanoparticulate zinc oxide, non-  
23 nanoscale zinc oxide and ionic zinc dosed as the chloride salt) influences soil bacterial community  
24 composition.

## 1 MATERIALS AND METHODS

### 2 *Chemicals*

3 Two zinc oxide particle forms were used in these experiments; uncoated 30 nm ZnO nanoparticles (ZnO  
4 NP) and 200 nm ZnO particles (ZnO P) (both Microniser Pth Ltd, Dandenong, Australia). Both the 30 nm  
5 ZnO NPs and 200 nm ZnO particles (hereafter referred to as ZnO NP and ZnO P) had no coatings or surface  
6 modification and were white odourless dry powders with close to spherical shaped particles. Primary  
7 particle sizes were verified by TEM (JEOL 2010 analytical TEM operating at 200kV). The hydrodynamic  
8 diameter of the particles in the stock suspension (nominal concentration 7.5 mg Zn/ml) and the zeta  
9 potential were determined by Dynamic Light Scattering (DLS) and Laser Doppler Electrophoresis (LDE)  
10 respectively using a Malvern Zetasizer Nano ZS. Full details and results of particle characterisation can be  
11 found in Heggelund *et al* (2014) which reports all the details of the exposures from which samples for  
12 microbial community analysis were taken (prior to the addition of earthworms). The ionic zinc reference  
13 material used in these experiments was zinc chloride (ZnCl<sub>2</sub>) (BHD Chemicals, Poole, UK).

14

### 15 *Soil and experimental setup*

16 The experimental procedures are fully described in a previous study investigating the toxicity of ZnO  
17 NPs and particles as well as ZnCl<sub>2</sub> to the earthworm *Eisenia fetida* in a single soil at varying pH levels  
18 (Heggelund et al. 2014). In brief, soil was collected from Wareham forest (Ordnance Survey Grid  
19 Reference: SU108058, Dorset, United Kingdom) and processed by removing large roots, homogenised  
20 and sieved through a 5-mm mesh followed by air drying (initial moisture content was approximately  
21 14% w/w). A summary of the soil properties can be found in Table S1. Soil pH was adjusted by the  
22 addition of 0.2%, 0.45% and 1% w/w calcium carbonate (CaCO<sub>3</sub>, Sigma Aldrich) to give pH values of  
23 4.5, 5.9 and 7.2 (range  $\pm$  0.3 pH units) referred to hereafter as the low-, medium- and high- pH soils,  
24 respectively. 450 g dry weight of the respective pH soil was added to the test container (1 L glass Kilner  
25 jars). The soils were spiked with ZnO NP, ZnO P and ZnCl<sub>2</sub>. The concentration range for the ZnO NP  
26 and ZnCl<sub>2</sub> treatments was 238, 381, 610, 976, 1520, and 2500 mg Zn/kg dw soil and the ZnO P treatment  
27 was 381, 976, and 2500 mg Zn/kg dw soil, with four replicates at each concentration at each of the three  
28 pH levels. An additional ten control soils with no zinc treatment were set up for each pH (total 210  
29 containers). The ZnO particles and zinc chloride were dispersed in soil solution at a high concentration  
30 before being added to the soil to give the appropriate zinc concentration in the soil. After spiking,  
31 water was added to the soil bringing the moisture content to 45% of the water holding capacity of the  
32 soil. Soils were incubated for seven days at 20 °C. At the end of the seven day period, soils were  
33 homogenised and 2 ml tubes were filled with soil from each container and stored at -80°C for  
34 molecular analysis.

### 1 *Pore water concentrations*

2 Zinc concentrations in the soil pore water were calculated in a parallel set of 200 g dry weight of soil  
3 of each pH, spiked with each form of zinc as described in Heggelund *et al.* (2014). This was done for  
4 all ZnO treatments and for three of the six ZnCl<sub>2</sub> treatments (381, 987 and 2500 mg Zn/kg). After  
5 dosing, the soils were saturated with MilliQ water and left for 10 days to equilibrate. Subsequently  
6 the soil pore water was extracted by centrifugation at 2482 g for two hours, and the supernatant  
7 centrifuged at 18,330 g for a further hour to minimise the number of remaining soil particles. The  
8 centrifuged soil pore water extracts were acidified by addition of 1 ml/ml 1 M nitric acid and zinc  
9 concentration in the pore water analysed by Atomic Absorption Spectroscopy (AAS) (Perkin Elmer  
10 1100B).

11

### 12 *Molecular assessment of bacterial composition*

13 Soil DNA extractions were carried out from 0.25 g of homogenised soil from all 210 containers. Total  
14 nucleic acids were extracted from the samples using the PowerSoil®-htp 96 Well Soil DNA Isolation Kit  
15 using standard manufacturer instructions (MO BIO Laboratories, Inc). Approximately 20–30 ng of  
16 purified template was used per PCR. For T-RFLP analysis, a ≈500 bp region of the 16S small subunit  
17 ribosomal RNA (SSU rRNA) was amplified using 6-FAM labelled forward primer 27F (3'-  
18 AGAGTTTGATCMTGGCTCAG) and reverse primer 536R (Suzuki *et al.* 1998). Thermal cycling conditions  
19 were as follows: initial denaturation at 94°C for 2 min followed by 30 thermal cycles of 94°C for 1 min,  
20 52°C for 1 min and 72°C for 3 min. The PCR products were gel-purified using a QIAquick Gel Extraction  
21 Kit (QIAGEN) and 20 µl of product was digested for 4 h at 37°C in a 30 ml total reaction volume using  
22 20 units restriction enzyme MspI (Promega). Digestion products (0.5 µl) were combined with  
23 denatured 0.5 µl LIZ600 size standard (Applied Biosystems) and 8.5 µl of Hi-Di formamide (Applied  
24 Biosystems) analysed on an Applied Biosystems 3730 DNA sequencer and the sizes of restriction  
25 fragments were calculated. Binning analysis was performed using Genemarker (Softgenetics).

26 A subset of samples representing two replicates of each control soil at each of the three pH levels, and  
27 one sample from each treatment (NS30, NS200, and ZnCl) at each pH level (4.8, 6.0 and 7.0) were  
28 characterised in more detail using 454 Next generation Sequencing (NGS). Briefly, pyrosequencing was  
29 carried out at Molecular Research LP, (Lubbock, Texas, USA). Microbial tag encoded FLX amplicon  
30 pyrosequencing (TEFAP) was conducted using 16S V1-V3 spanning primers Gray28F  
31 GAGTTTGATCNTGGCTCAG and Gray519r GTNTTACNGCGGCKGCTG. Generation of the sequencing  
32 library used a one-step PCR with for 30 cycles, a mixture of Hot Start and HotStart high fidelity taq



1 polymerases, and amplicons originating and extending from the forward primers. Pyrosequencing  
2 analyses utilized Roche 454 FLX instrument with Titanium reagents.

3

#### 4 *Data analysis*

5 All statistical analyses were carried out in R (v.3.0.1) (R Core Team 2013) using the package ‘Vegan’  
6 v2.0-10 (Oksanen et al. 2013). To visualise the relationship between TRFLP profiles from soils with  
7 different treatments we used Non-Metric Multidimensional Scaling (NMDS) using the ‘metaMDS’  
8 function, based on dissimilarities calculated with the Bray–Curtis index. Proximity of the points on the  
9 NMDS plot characterises the between sample similarity of bacterial communities, with points close  
10 together representing similar communities and points further apart representing dissimilar bacterial  
11 communities. NMDS was initially used to examine the overall effect of pH modification on soil bacterial  
12 communities. Additionally, bacterial diversity, as measured using Shannon’s diversity Index ( $H'$ ) on the  
13 TRFLP peaks was examined for each pH class using the control samples. Bray-Curtis distances were  
14 plotted against zinc dose to examine community dose response relationships. The function ‘Adonis’,  
15 which uses a permutational analysis of sums of squares using semi-metric and metric distance  
16 matrices, was used to test for statistically significant differences between the bacterial communities  
17 in the control soils and treated soils. The F-statistic was plotted on a scatterplot to visualise the scale  
18 of the effect against the dose of ionic, particulate and nano zinc.

19

20 For the 454 data, sequence processing, denoising, error checking and clustering was carried out as  
21 described in Read *et al.* (2015). The taxonomy of OTUs was determined by RDP Classifier with 80%  
22 bootstrapping classification confidence (Wang et al. 2007) using the Greengenes Oct 2012 database  
23 (McDonald et al. 2012).

24

## 25 RESULTS AND DISCUSSION

26

### 27 *Material characterisation*

28 Material characterisation TEM analysis indicated that the particles were spherical and relatively  
29 monodisperse in the case of ZnO NPs. The non-nanoscale material (ZnO P) contained a higher  
30 proportion of faceted rod-shaped material. Images used for characterisation of the particles in distilled  
31 water and dosing solution are presented in Heggelund *et al.* (2014). The average primary particle  
32 diameters of the ZnO material batches as measured by TEM were  $29.8 \text{ nm} \pm 9.4$  (mean  $\pm$  standard  
33 deviation) for the ZnO NPs and  $300 \text{ nm} \pm 164$  (length) and  $188 \text{ nm} \pm 102$  (width) for the non-nanoscale  
34 ZnO (Waalewijn-Kool et al. 2012).

## 1 *Effect of soil pH on bacterial community composition*

2 Soil pH has previously been shown to strongly correlate with bacterial community composition and  
3 diversity, with a positive relationship between pH and diversity observed in both natural soils (Fierer  
4 & Jackson 2006, Griffiths et al. 2011) and those with artificially modified pH ranges (Bartram et al.  
5 2014, Rousk et al. 2010). Although the exact mechanism behind this is not known, it has been  
6 speculated to be due to bacterial taxa exhibiting relatively narrow growth tolerances to pH (Rousk et  
7 al. 2010). Despite this appearing to be a fundamental rule determining microbial composition in soils  
8 worldwide, little research has been done comparing resistance and resilience of bacterial communities  
9 across this range. In the current study, modification of soil pH resulted in a predictable shift in  
10 community composition, with an increase in Proteobacteria and reduction in Acidobacteria with  
11 increasing pH (Figure 1). An NMDS ordination (Figure 2A) of the control (no zinc addition) communities  
12 showed that increasing soil pH through the addition of calcium carbonate (from a starting pH of  $\approx 4.0$ )  
13 to values of 4.5, 5.9 and 7.2, caused shifts in bacterial community composition along Axis 1. This was  
14 accompanied by changes in the diversity of TRFLP peaks (Shannon H' Index), where increasing the soil  
15 pH resulted in increased measured diversity (Figure 2B), in agreement with previous studies (Fierer &  
16 Jackson 2006, Griffiths et al. 2011).

17 Having established three pH modified bacteria communities, we wished to examine two aspects of  
18 the response of soil bacterial communities to zinc exposure; 1) the interaction between soil pH and  
19 zinc exposure on soil bacterial communities, and; 2) whether zinc form (nanoparticulate non-  
20 nanoscale and ionic) influences the response of soil bacterial communities to exposure.

21 The relationship between bacterial communities in each treatment (ZnO NP, ZnO P and ZnCl<sub>2</sub>) in  
22 relation to treatment dose and soil pH was examined using a matrix of NMDS ordination plots (Figure  
23 3). Exposure to zinc in all forms caused changes in the bacterial community when compared to the  
24 control samples, illustrated by the increased distance between the treatment and control points.  
25 Generally, there was a higher change in zinc exposed communities in low and medium pH soils (pH 4.5  
26 and 5.9) versus the high pH soil (pH 7.2). This change is quantified in Figure 4, which shows the  
27 between group (control and treatment dose) F-values from an Adonis analysis. Higher F-values at  
28 higher dose concentrations indicate a larger effect of zinc on the bacterial community composition. A  
29 dose-response can be observed in the low and medium pH soils, where higher doses generally result  
30 in higher changes in community composition. However, in the high pH soil this relationship breaks  
31 down, and the community is relatively insensitive to zinc addition. These results are also reflected in  
32 an examination of community dissimilarity using Bray-Curtis distances, where increasing dose  
33 increases dissimilarity between samples (Supplementary figure S1). Possible explanations for this

1 pattern are twofold. Firstly there is a known interaction between pH and nanoparticle dissolution  
2 (Zhang et al. 2010). Our measurements of soil pore water zinc concentration indicated that pH controls  
3 zinc dissolution and availability, with zinc pore water concentrations being higher in lower pH soils  
4 (Figure 5). This relationship has been observed to drive similar differences in effects in previous  
5 nanotoxicity studies with invertebrates (Waalewijn-Kool et al. 2013) and is due to more acidic  
6 conditions generally favouring dissolution of ZnO and release of Zn ions (Miao et al. 2010). However,  
7 the availability of zinc (as measured by pore water concentration) does not completely explain the  
8 patterns in bacterial community composition. For example, zinc pore water concentrations in the ZnO  
9 NP treatment did not vary between the pH 5.9 and pH 7.2 treatments (Figure 5A), but there was a  
10 much larger change in community composition with increasing zinc dosage in the pH 5.9 soil compared  
11 to the pH 7.2 soil (Figure 4A). This means there may be a specific response to nanoparticles, and that  
12 this is linked to pH in a manner that does not just involve particle dissolution. One explanation is that  
13 pH has previously been observed to influence nanoparticle zeta potential and this has been related to  
14 nanotoxicity due to interactions with cell membranes (Berg et al. 2009, Cho et al. 2012, Schwegmann  
15 et al. 2010).

16 The second explanation is that pH-specific bacterial communities may vary in their resistance to zinc,  
17 with low pH communities being inherently more susceptible to Zn exposure than high pH  
18 communities. Our results indicated that the lower pH communities had a lower diversity, as measured  
19 using Shannon's H index on the TRFLP peaks. Although there is some debate as the validity of diversity  
20 indices as an absolute measure of diversity when used on TRFP data (Blackwood et al. 2007), these pH  
21 driven patterns are in agreement with large scale sequencing based assessments of soil bacteria  
22 diversity (Fierer & Jackson 2006). If the lower pH soils do have a lower diversity, it may explain reduced  
23 resilience to disturbance, based on lower levels of functional and taxonomic redundancy. We are  
24 unaware of any studies that address this question directly, although bacterial communities across soil  
25 pH ranges have previously been shown to vary in their functionality, including growth and rates of  
26 biogeochemical cycles (Fernandez-Calvino et al. 2011, Kemmitt et al. 2006). The design of this study,  
27 and the nature of zinc solubility in soil means that it is not possible to unravel the relative contribution  
28 of the role of pH on zinc dissolution and bacterial community structure, but it would make a valuable  
29 topic for future research.

### 30 *Effect of Zinc form on microbial communities*

31 Finally, we wished to address the question as to whether there was a specific microbial response  
32 related to the form of zinc (ionic, nanoparticulate or particulate). Previous studies have highlighted  
33 that the physical characteristics of nanoparticles have a major influence on toxicity, mainly focussing

1 on NP size (Coradeghini et al. 2013, Gliga et al. 2014), but also shape (Hua et al. 2014) and surface  
2 chemistry (Albanese et al. 2012). Size-dependent toxicity is thought to occur due to direct interactions  
3 between nanoparticles and cells, with smaller particles allowing for a higher contact rate with cell  
4 membranes (Nair et al. 2009), but also higher rates of dissolution in the environment and release of  
5 ionic metal (Ma et al. 2012). Additionally, there has been debate over whether nanoparticles are  
6 inherently toxic or that toxic mechanisms are solely linked to the release of the ionic metal.

7 In the present study, we observed ionic and particulate specific responses in soil microbes across all  
8 doses (Supplementary Figure S2), although this was the most pronounced at the highest dose of 2500  
9 mg Zn/kg dw soil (Figure 6). As before, it can be seen that all forms of Zn caused the communities to  
10 change in structure compared to the control soils, regardless of soil pH. However, the bacterial  
11 communities after exposure to particulate zinc (either nano or micro-scale) clustered together, and  
12 differed to those in the dissolved zinc salt treatments. One explanation is that the addition of ionic  
13 zinc in the form of zinc chloride resulted in far higher soil pore water concentrations (ranging from  
14 90.4 – 809 mg/L) compared to either of the particulate zinc treatments (6.93 – 14.8 mg/L) (Figure 5).  
15 The results also showed that there was little difference in the nanoparticulate (NS30) and micro-scale  
16 (NS200) zinc treatments at the highest doses, indicating that there is no clear evidence of toxicity  
17 directly related to the presence of nanoparticles rather than non-nanoparticulate zinc.

## 18 *Conclusions*

19 Determining nanotoxicity in soils is complex, largely because of the numerous ways in which  
20 nanoparticles can interact with their environment, including through dissolution, agglomeration and  
21 the addition of environmentally obtained surface coatings, all of which have been shown to modify  
22 toxicity in model systems (Albanese et al. 2012). This is further complicated by the fact that  
23 interactions with organisms can further modify the properties, and hence toxicity of nanoparticles in  
24 the environment. For example, many bacteria produces extracellular polymeric substances (EPS) and  
25 exudates that have been shown to modify nanoparticle behaviour (Duster & Fein 2014, Kroll et al.  
26 2014). Our study was designed to investigate the interactions with pristine nanoparticles added direct  
27 to soil, a situation that is unlikely unless due to industrial contamination or spillage. A far more likely  
28 route of nanoparticle transmission is through sewage works, and the addition of sewage sludge to  
29 soils. Nanoparticle have been shown to exhibit a wide range of chemical transformations in this  
30 scenario (Lombi et al. 2013, Ma et al. 2014) changing many of the properties, such as size, shape and  
31 surface coatings, that determine toxicity. Because of this, further work should focus on  
32 environmentally relevant routes of exposure and determine the impacts on microbial communities  
33 over a range of timescales.

1

## 2 Acknowledgements

3 The NanoFATE Project CP-FP 247739 (2010-2014) under the 7th Framework Programme of the  
4 European Commission (FP7-NMP-ENV-2009, Theme 4) coordinated by C. Svendsen; [www.nanofate.eu](http://www.nanofate.eu)  
5 and DEFRA project CB0460 are acknowledged for financial support. Dr. M. Diez-Ortiz was supported  
6 by a Marie Curie Intra-European Fellowship within the 7th European Community Framework  
7 Programme (call reference FP7-PEOPLE-2010-IEF, 273207 Nano-Ecotoxicity).

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- 8

1 Figure legends

2 **Figure 1.** Phylum-level bar chart showing the representative bacterial composition of the control (no  
3 zinc) and treated soils exposed 2500 mg Zn/kg dw soil of 30 nm nanoparticulate zinc (ZnO NP), non-  
4 nanoscale particulate zinc (ZnO P) and Zinc chloride (ZnCl<sub>2</sub>) at each pH (4.5, 5.9 and 7.2) from 454 16S  
5 rRNA gene sequencing.

6

7 **Figure 2.** A: The relationship between TRFLP profiles from control soils at three pH levels in a Non  
8 metric Multidimensional Scaling (NMDS) plot (A). Colours show the three different soil pH groups (red  
9 pH 4.5, orange pH 5.9, green pH 7.2). The Shannon diversity (H) of the TRFLP profiles from control soils  
10 across each of the three pH treatments (B).

11

12 **Figure 3.** Non Metric Multidimensional Scaling (NMDS) plots showing the relationship between the  
13 microbial community composition in the ZnO NP (NS30), ZnO P (NS200) and Zinc chloride (ZnCl<sub>2</sub>)  
14 treatments in soils of pH 4.5, 5.9, and 7.2.

15

16 **Figure 4.** Line charts showing the relationship between the Adonis F-test values of each dose group  
17 compared to the control for ZnO NP (NS30) (A), ZnO P (NS200) (B) and zinc chloride (ZnCl<sub>2</sub>) (C). A higher  
18 F-test value indicates a higher level of significance in the difference between the groups.

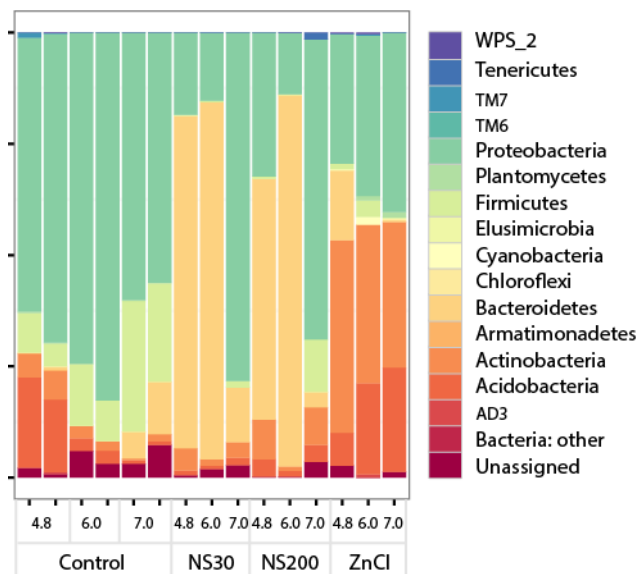
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20 **Figure 5.** Dosed zinc concentration versus measured zinc concentration in pore water at each soil pH  
21 for zinc chloride (ZnCl<sub>2</sub>) (A), ZnO NP (NS30) (B) and ZnO P (NS200) (C).

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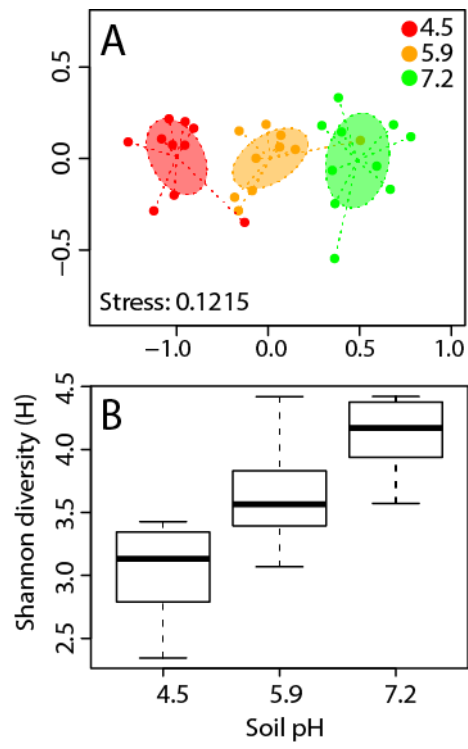
23 **Figure 6.** Non Metric Multidimensional Scaling (NMDS) plots showing the relationship between the  
24 microbial communities from TRFLP profiles for each treatment (ZnO NP (NS30), ZnO P (NS200) and  
25 Zinc chloride (ZnCl<sub>2</sub>)) at the highest dose (2500 mg Zn/kg dw soil) in soils of pH 4.5 (A), 5.9 (B), and 7.2  
26 (C).

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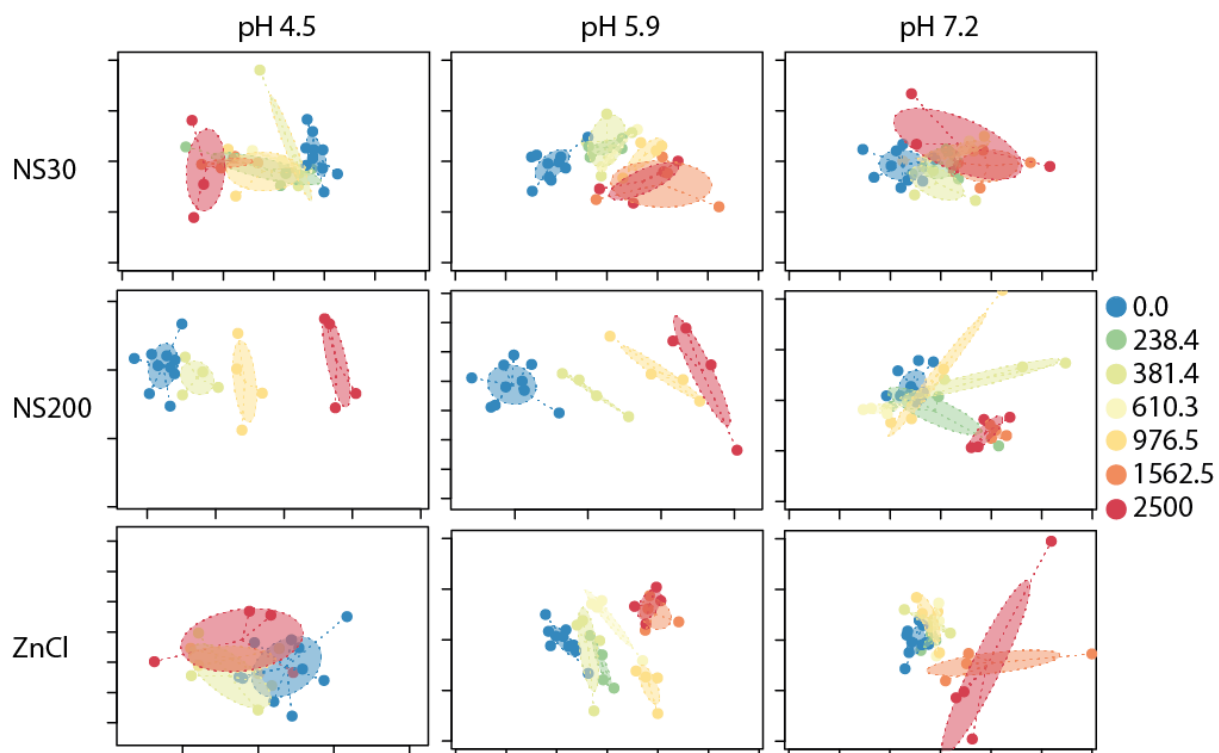
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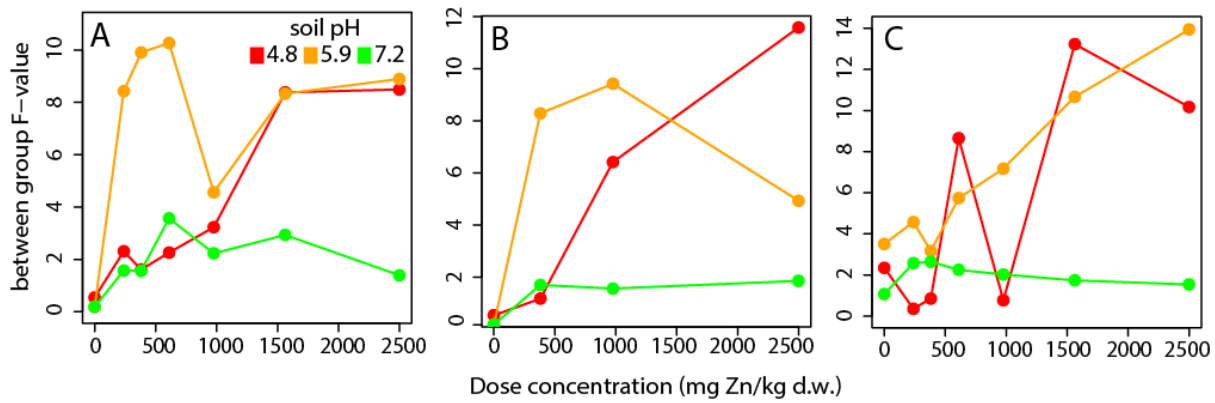


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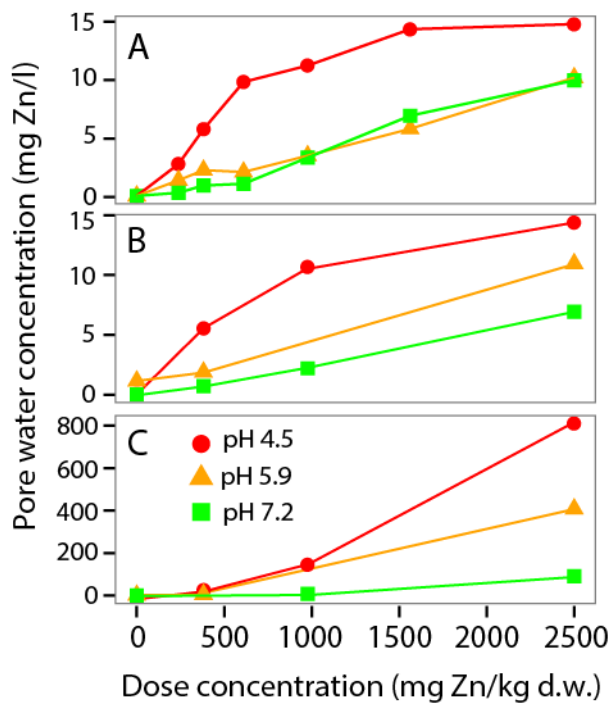


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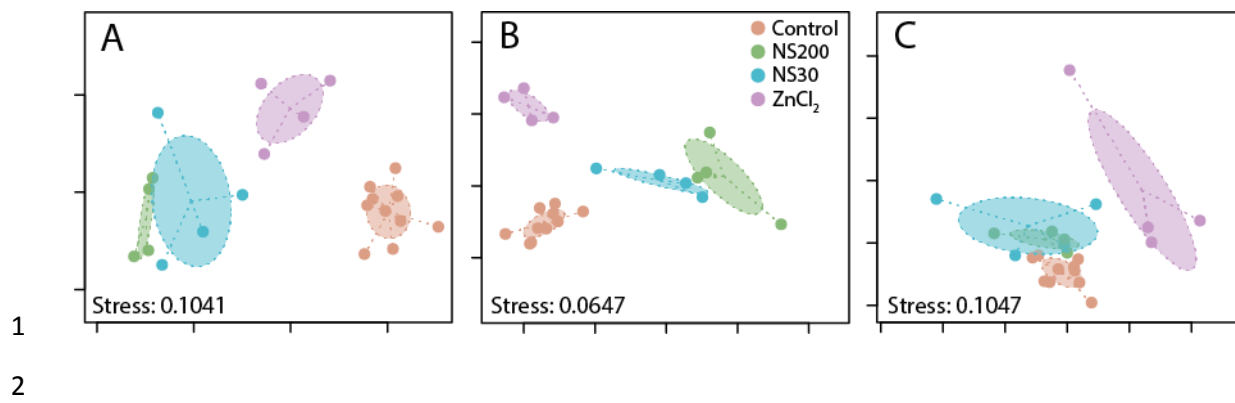
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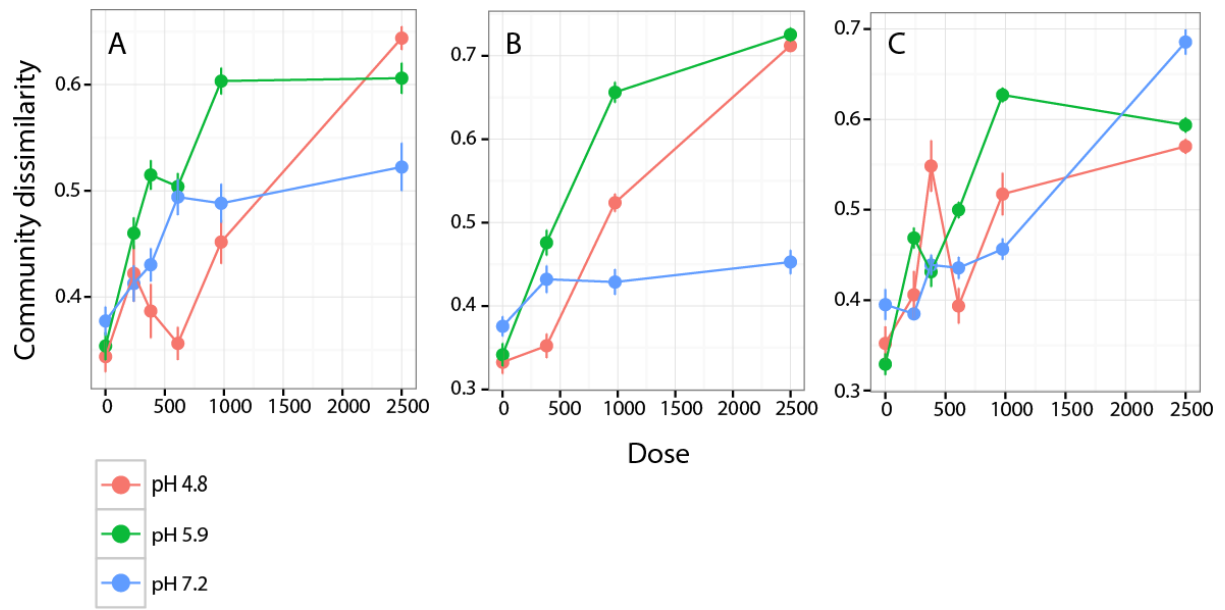




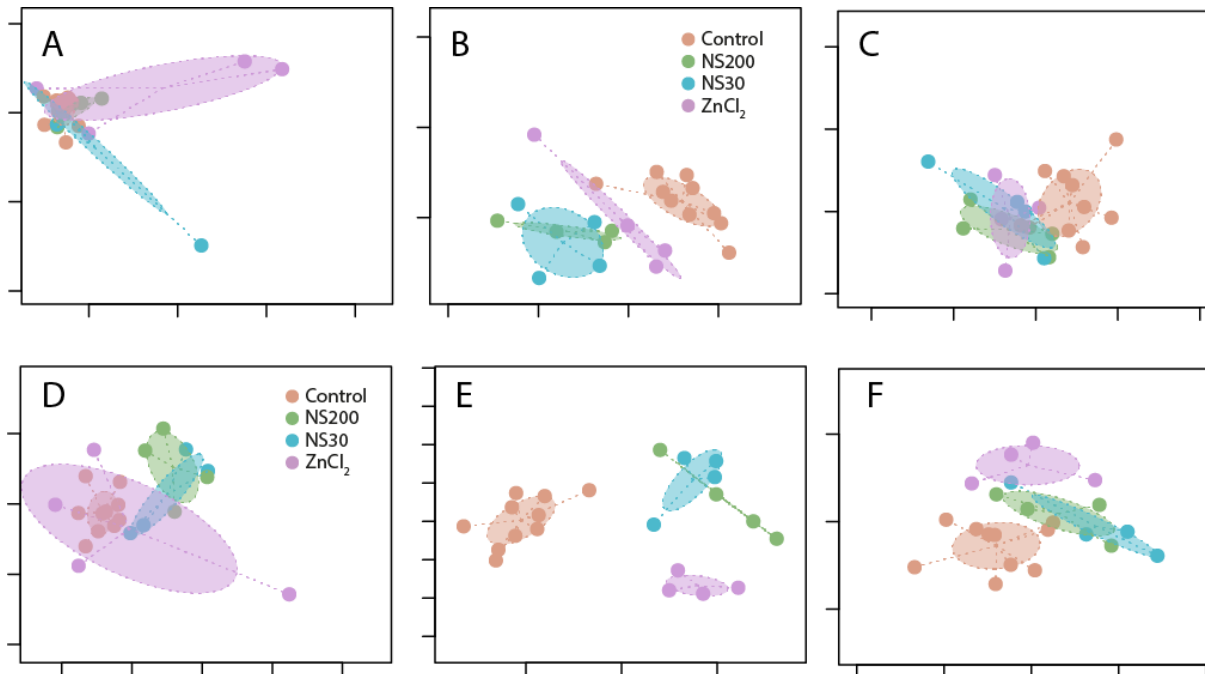
1 **Table S1.** Summary of the soil properties of the unchanged Dorset soil prior to pH amendment. Dorset  
 2 soil texture, pH (in water and 0.01M CaCl<sub>2</sub>), conductivity, total carbon (% Total C) and nitrogen (%  
 3 Total N), cation exchange capacity (CEC), percent base saturation (% BS), major cations concentrations  
 4 (Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup> and Na<sup>+</sup>) along with oxalate (% Fe<sub>ox</sub>, % Al<sub>ox</sub>) and citrate-bicarbonate-dithionite (%  
 5 Fe<sub>CBD</sub> and % Al<sub>CBD</sub>) extractable iron and aluminium are shown (Heggelund et al 2014)  
 6

	<b>% Coarse sand</b>	<b>% Fine sand</b>	<b>% Clay</b>	<b>% Silt</b>	<b>% Total C</b>	<b>% Total N</b>
<b>Texture, total carbon &amp; nitrogen</b>	51.5	40.2	3.5	4.7	4	0.1
	<b>pH<sub>H2O</sub></b>	<b>pH<sub>CaCl2</sub></b>	<b>Conductivity (µS)</b>			
<b>pH and conductivity</b>	4.2	3.1	422			
	<b>Ca</b>	<b>Mg</b>	<b>K</b>	<b>Na</b>	<b>CEC</b>	<b>% BS</b>
<b>Exchangeable cations (cmol(+)/kg)</b>	1.4	0.6	0.1	0.1	5.4	41
	<b>% Fe<sub>ox</sub></b>	<b>% Al<sub>ox</sub></b>	<b>% Fe<sub>CBD</sub></b>	<b>% Al<sub>CBD</sub></b>		
<b>Extractable Fe &amp; Al</b>	0.04	0.03	0.11	0.03		

1 **Supplementary Figure S1.** Community dissimilarity (Bay Curtis distance) between samples for zinc  
 2 treatments containing nanoparticulate zinc (A), non-nanoscale zinc particles (B) and zinc chloride (C).  
 3 Points and lines represent treatments in soils of different pH levels.  
 4



1 **Supplementary figure S2.** Non Metric Multidimensional Scaling (NMDS) plots showing the relationship  
2 between the microbial communities from TRFLP profiles for nanoparticulate zinc (ZnO NP), non-  
3 nanoscale zinc particles (ZnO P) and zinc chloride (ZnCl<sub>2</sub>) at zinc doses of 381.47 (A-C) and 976.56 (D-  
4 F) mg Zn / kg DW soil in soils of pH 4.5 (A, D), 5.9 (B, E), and 7.2 (C, F).



5  
6