2012-12

Effects of Salinity on the Toxicity of Ionic Silver and Ag-PVP Nanoparticles to *Tisbe Battagliai* and *Ceramecium Tenuicorne*

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Recommended Citation

Macken, A., Byrne, H.J., Thomas, K.V., (2012) Effects of salinity on the toxicity of ionic silver and Ag-PVP nanoparticles to *Tisbe battagliai* and *Ceramecium tenuicorne*, *Ecotoxicology and Environmental Safety*, 86, 101-110 (2012) doi:10.1016/j.ecoenv.2012.08.025

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Funder: HEA PRTL Cycle 4 INSPIRE

1 **Effects of salinity on the toxicity of ionic silver and Ag-PVP nanoparticles to *Tisbe***
2 ***battagliai* and *Ceramium tenuicorne***

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Abstract

The toxic effects of polyvinylpyrrolidone (PVP) coated silver nanoparticles (Ag-NP_{PVP}) and ionic Ag, to *Tisbe battagliai* (Tb) and *Ceramium tenuicorne* (Ct) were investigated and the usefulness of standardised marine guidelines for ENP risk assessment were assessed. The toxicity of Ag-NP_{PVP} [CtEC₅₀ = 26.6 µg/L, TbEC₅₀ = 7.9 µg/L] and Ag⁺ [CtEC₅₀ = 2312.2µg/L, Tb EC₅₀ = 90.9 µg/L] to both test species differed, with the silver ENPs being more toxic. In contrast to Ag⁺ the toxicity of Ag-NP_{PVP} increased significantly with increasing salinity, however, after thorough characterisation it was not possible to correlate the behaviour of the particles with an increase in toxicity and salinity. The results suggest that the observed toxicity is being elicited by the free ionic silver complexing in solution and also from an unknown potential particle related effect.

Keywords: silver nanoparticles, marine ecotoxicity, salinity, *Tisbe battagliai*, *Ceramium tenuicorne*

1 Introduction

Silver nanoparticles are one of the most widely used engineered nanomaterials (NMs) employed in commercial applications and their wide use in consumer products has led to them becoming a potentially important factor in environmental risk assessment (ERA) and monitoring (Klaine et al., 2008; Meyer et al., 2010). Naturally occurring nanoparticles have always existed in the environment without documented adverse effects (Murr et al., 2004; Handy et al., 2008). Despite this, the risks associated with engineered nanoparticles (ENPs) in the environment are becoming a significant concern (Moore, 2006; Ju-Nam and Lead, 2008; Tiede et al., 2009).

The ERA of NMs is a relatively new field and many questions about appropriate assessment methods exist (Tiede et al., 2009; Bhatt and Tripathi, 2011). At present there is inadequate information on the levels of nanomaterials in the environment (Paterson et al., 2011). Most work to date has focused on laboratory based evaluation of risk, where few interactions between the ENPs and natural environmental constituents occur. It is only in recent years that scientists have started to incorporate realistic environmental exposures into their laboratory work (Diegoli et al., 2008; Van Hoecke et al. 2011). Despite recent changes to a more “environmentally relevant” approach in ENP ecotoxicity and fate assessment, it can still be stated that, at present there exists no robust, reliable method for the quantification or assessment of ENPs in the environment.

Most ENP ecotoxicological assessments have, up to now, focused on the potential effects of ENPs in freshwater environments covering a wide array of species including fish (Farkas et al., 2011; Farkas et al., 2010; Bilberg et al., 2010), daphnids (Naddy et al., 2007), nematodes (Meyer et al., 2010), chironomids (Nair et al., 2011) and microalgae (Petit et al., 2010). Due to the increased use of silver ENPs in consumer products there is a likelihood that they will also end up in marine, estuarine and coastal environments. The possible adverse

effects elicited by these compounds must therefore be assessed employing suitable marine test systems. A small number of studies employing marine species in the laboratory have been performed (Galloway et al, 2010; Ringwood et al. 2010; Canesi et al., 2010), but there is still a paucity of available data. Fabrega et al (2011) demonstrated that biofilm succession was impeded on silver ENP treated biofilms, which in turn affected the abundance of major bacterial groups that may cause further long term effect on the marine bacterial community. In contrast Miao et al (2009) observed that silver ENPs formed non-toxic aggregates ($> 0.22 \mu\text{m}$) in seawater. Assessing the effects of ENP exposure to marine organisms poses new and complex issues, hence, the importance of a realistic environmental assessment of both the behaviour and effects needs to be emphasised.

The high ionic strength of seawater is likely to affect the behaviour of ENPs and may prevent the introduced ENPs from becoming completely dispersed within the marine and estuarine environment due to agglomeration (Gebauer and Treuel, 2011). There have been some recent studies investigating the effects of salinity on ENPs revealing that only minute changes in salinity may drastically affect the behaviour, effects and fate of ENPs (Matson, 2009). In the case of an estuarine environment, it is of even more importance that the behaviour of ENPs and the effects of salinity be understood. Estuaries are constantly changing and highly dynamic environments, in which organisms have to adapt to and tolerate rapid changes in physico-chemical conditions over short periods of time. Estuaries and coastal environments are also the most likely of the marine ecosystems to be impacted by potential ENP contamination (Kaegi et al., 2008).

Silver ENPs are of particular interest in the marine environment, not only because of their abundant use and potential entry into this ecosystem, but also because the speciation behaviour of silver in saline waters increases bioavailability (Luoma et al., 1995). Upon entering the marine environment, silver speciation becomes a significant factor (Cowan et al.,

1985). Therefore, in the marine and estuarine environment, silver is unlikely to remain in the form in which it was introduced. Another potential effect on the behaviour of ENPs in saline media, is the fact that their tendency to aggregate/agglomerate may increase with increased salinity.

The primary production of micro and macro algal species in the marine environment is of paramount importance. Effects on primary production within an ecosystem could lead to devastating effects on ecosystem functioning. Sub-lethal effects on these organisms may also lead to transfer and biomagnification of metal contaminants up the food chain (Quigg, 2008). At present, research on the interactions and toxicity of ENPs to macrophytes is lacking for both the terrestrial and aquatic ecosystems. Due to this lack of information on higher plants, *Ceramium tenuicorne*, a marine rhodophyte, was selected for use in this study. *C. tenuicorne* was selected as it represents a primary producer in both the marine and estuarine environment (i.e. euryhaline species) and could be used to assay the ENPs under a wide range of salinities. *Ceramium* species are found world-wide and are particularly abundant in temperate climates. The marine clone employed in this study originated from the Oslofjord (20 – 25 ‰), an inlet in the south-east of Norway, and has been maintained as a laboratory culture for over 30 years and can be adapted to a wide range of environmentally relevant salinities (ISO, 2010).

In order to augment the multi-trophic nature of this study, a marine harpacticoid copepod *Tisbe battagliai*, was incorporated into the testing. *T. battagliai* represents a primary consumer within the marine food web and is of environmental significance because, along with nematodes, copepods are the most abundant multicellular organisms on earth (Humes, 1994). It is also a standard test species in regulatory testing and employed in ERA. To the best of our knowledge, no data regarding the effects of ENPs to *T. battagliai* and *C. tenuicorne* exist in the literature. Therefore, the information gained in this work is considered

to contribute to the knowledge gap in understanding the effects of ENPs to these marine species.

Although there are many studies in the literature on the ecotoxicological assessment of ENPs, there is no standardisation in the methodologies used between different laboratories. Significant factors, such as sample preparation and ENP characterisation, vary significantly between laboratories. This variation leads to considerable differences in results for similar particles and incomparable inter-laboratory data, for both ecotoxicological tests and characterisation (Roebben et al., 2011, Petersen and Henry, 2012). The applicability of standard guidelines for the assessment of ENPs in the environment is of great interest at present. Several major international standards organisations (e.g. OECD and ISO) are currently investigating the applicability of the standard methods for use in ERA and regulation of ENPs. Due to the potential for more rigorous ERA of ENPs in the future, where possible the studies performed in this work were performed in line with standard regulatory guidelines (ISO 14669, ISO 10710) and according to GLP (Good Laboratory Practice, NIVA has a GLP certified ecotoxicological laboratory), any deviations are described in full.

The main objectives of this study were to: (1) investigate the toxicity of silver nitrate (AgNO_3) and polyvinylpyrrolidone coated silver nanoparticles ($\text{Ag-NP}_{\text{PVP}}$) to two marine species and to fully support all ecotoxicological studies with a thorough characterisation of the ENPs in all environmental media; (2) to investigate species specific differences in sensitivity; (3) evaluate the potential modifying effects of varying salinity on the toxicity of AgNO_3 and $\text{Ag-NP}_{\text{PVP}}$ to *C. tenuicorne* and (4) to place the results in an environmentally relevant context and to discuss the applicability of these standardised test guidelines for use in the regulatory assessment of engineered NPs within the marine and estuarine environment.

2 Materials and Methods

2.1. Nanoparticles and test chemicals

The silver nanoparticles employed in this study were coated with polyvinylpyrrolidone (PVP). These PVP (Ag-NP_{PVP}) capped ENPs were kindly prepared and provided in a solution of MilliQ water by the University of Manchester, UK. Post analysis of the stock solution by ICP-MS, indicated that the concentration was 348 mg/L, based on a measurement of total silver.

Four different types of media were used in these studies: non-enriched natural seawater (NSW) for the *T. battagliai* assays and an enriched algal media for *C. tenuicorne* assays (Eklund, 2005, ISO, 2010), at three different salinities (to be referred to as CT10 ‰, CT20 ‰, and CT30 ‰). Natural seawater, collected from 60 m depth at Solbergstrand, Norway (ca. 34 ‰), was employed in the preparation of all media. All media were filtered to 0.2 µm prior to use. Test solutions of ENPs for use in the ecotoxicity assessments were prepared by diluting the stock solution in the appropriate saline media. Dilutions were made directly from the stock solution, where possible, in order to reduce the effects of potential aggregation/agglomeration during serial dilution. All test solutions were stirred for 3 hours at 100 rpm prior to characterisation and use in ecotoxicology tests.

The reference chemical, analytical grade potassium dichromate (K₂Cr₂O₇; CAS Registry No 778-50-9) was obtained from Sigma Aldrich. Polyvinylpyrrolidone (PVP; CAS Registry No 9003-39-8) also obtained from Sigma Aldrich, was employed as a control for the nanoparticle capping agent.

2.2. Nanoparticle characterisation

There are many techniques for measuring the size of nanoparticles, all of which are based on fundamentally different scientific principles. It has previously been reported that no single technique is sufficient to accurately estimate the size of nanoparticles in all media types (Domingos et al., 2009 Scalf and West, 2006). Each individual technique has intrinsic biases and artefacts produced during sample preparation, or from instrumental interference (Domingos et al. 2009). Therefore, in this study, a battery of different characterisation techniques was employed to investigate the size and behaviour of the Ag-NP_{PVP} in saline media.

2.2.1. Dynamic Light Scattering (DLS)

Dynamic light scattering (DLS) was used to measure the hydrodynamic diameter (Z-average), the intensity based size distribution and the zeta potential of Ag-NP_{PVP} in MilliQ water and all 4 media types. The particle size distributions (PSD) of the Ag-NP_{PVP} were analysed using a Malvern Instruments Zetasizer Nano Series (Malvern Instruments, UK). Concentrations of 100 µg/L were prepared in all media and in MilliQ water. Samples were analysed at 20 °C. Typically, 1 mL of solution was placed into a disposable sizing cuvette and samples for all media were prepared and analysed in triplicate. Six measurements of each triplicate sample were taken for all media.

Using the same Malvern Instrument, zeta potential measurements were also performed on the same solutions described above. Folded capillary cells with ca. 3 mL of each solution were employed. Measurements were conducted at 20 °C, using a concentration of 100 µg/L silver ENP and 6 measurements on triplicate samples were made.

2.2.2. Atomic Force Microscopy (AFM)

Atomic force microscopy (AFM) was used to resolve particles and groups of particles by offering visualisation in three dimensions. Samples of Ag-NP_{PVP} were prepared in all media types and MilliQ water at a concentration of 10000 µg/L. Samples were drop cast onto silicon wafers which were subsequently rinsed with de-ionised water and imaged in tapping mode with a high performance AFM microscope (Asylum MFP-3D-BIO™).

2.2.3. Transmission Electron Microscopy (TEM)

Solutions of the highest working concentrations of silver ENPs were prepared in all test media. The size and shape of the particles were determined by TEM. The ENP suspensions were prepared on the day of analysis in appropriate test media and applied to carbon coated grids (5 µl pipetted directly onto the grids). The grids were allowed to dry and images were captured with a Philips CM 100 electron microscope. Using image analysis software (Digimizer Version 3.4.1.0) to analyse micrograph images, 100 ENPs were measured and the mean particle size and PSD of Ag-NP_{PVP} in all media described.

2.3. Ecotoxicity testing

2.3.1. *Tisbe battagliai* culturing and testing

T. battagliai were cultured in the laboratory at NIVA in accordance with standard procedures (ISO, 1999) using NSW of ca. 34 %. Cultures were fed a single algal diet consisting of *Rhodomonas baltica* once per week (additional feeding may be required depending on the ability of the animals to clear the food) during culture water renewal.

T. battagliai toxicity tests were conducted with slight modifications according to the ISO method (ISO, 1999). Toxicity tests were conducted with copepodids 6 ± 2 days old. The bioassays were performed in 12-well polystyrene tissue culture plates (NUNC®) that had been pre-treated overnight with appropriate exposure solutions. After initial range finding studies, both Ag-NP_{PVP} and AgNO₃ were tested over the following range 1, 3.2, 5.6, 10, 32, 56, 100 µg/L (7 concentrations plus a negative control [NSW]). Controls for PVP were run alongside to confirm that any observed toxicological effects with Ag-NP_{PVP} were not due to the presence of the capping agent. All concentrations were run in quadruplicate (2.5 mL of exposure solution per replicate with five test organisms per replicate, a total of 20 animals per test concentration). Test plates were incubated in a temperature controlled room at 20 ± 2 °C and a 16:8 hour light:dark photoperiod. Experiments were performed on three independent occasions.

A reference substance, potassium dichromate (K₂Cr₂O₇), was run alongside the tests in order to verify the sensitivity of the copepods and to ensure the test conditions were reliable (ISO, 1999). At test initiation and termination, dissolved oxygen concentrations, pH and salinity were measured in the control, top and bottom concentrations for all tests. Control species mortality of less than or equal to 10 % was required as a further validation parameter.

2.3.2 *Ceramium tenuicorne* adaptation

A sample of *C. tenuicorne* (ITM, Stockholm) was supplied for use in this study (20 ‰), with a pre-testing adaptation period, to ensure good growth of the clone at all proposed test salinities.

Adaptation of the plants to 10 and 30 ‰ was conducted over several weeks prior to testing. This was done by successively transferring the algae every three days to an

increase/decrease of approximately 3 %. The algae were then cultivated for at least two weeks in the final test salinity prior to test initiation.

Prior to testing, the growth rate of the algae at the varying test salinities was determined to ensure sufficient growth during the test period. This was carried out by conducting a 7 day growth experiment according to the ISO guideline (ISO, 2010) with exponentially growing female tips (0.6 – 1.2 mm start length) in all media without test chemical. All media growth experiments were conducted in triplicate and measurements, at test initiation and termination, were taken using an Olympus 1x71 fluorescent microscope (without fluorescent mode) (Tokyo, Japan) and Olympus Soft Imaging system: Cell[^]D, Version 3.2 (Build 1700).

2.3.3 *Ceramium tenuicorne* silver toxicity and salinity effects on growth rate

The toxicity of Ag-NP_{PVP} and AgNO₃ as well as the effects of salinity on these toxicities, to the female plants, were studied using pre-adapted *C. tenuicorne* (section 2.3.2).

After initial range finding studies, definitive toxicity tests at all salinities were conducted within the range 1, 5, 10, 50, 100, 500, 1000, 5000, 10000 µg/L (9 concentrations and appropriate negative controls), for both Ag-NP_{PVP} and AgNO₃. As with the *T. battaglai* assay, a PVP control was incorporated into the test design. All toxicity testing was conducted according to modified standard methods (Eklund, 2005; ISO, 2010). Due to the small amount of test material available for Ag-NP_{PVP}, exposure volumes were reduced to 2.5 mL per replicate (4 replicates/concentration). 12-well pre-treated tissue culture plates (NUNC[®]) were used for the exposures. All concentrations were run in quadruplicate with two algal plants per replicate. Microplates were incubated for seven days at a temperature of 22 ± 2 °C, a light intensity of 70 ± 10 % µmol m⁻² s⁻¹ and a continuous light regime. In order to prevent evaporation (due to the small volume size) plates were sealed with parafilm. Experiments

were conducted for all salinities, for both silver compounds, on three independent occasions to ensure reproducibility and increase statistical power. A reference chemical (potassium dichromate) was run alongside to ensure performance and reliability of test methods. At test initiation and termination, dissolved oxygen concentrations, pH and salinity were measured in the control, top and bottom concentrations.

2.4 Statistical analysis

The EC₁₀/EC₅₀ (concentration that elicits an estimated 10 %/50 % toxic effect (i.e. growth inhibition, mortality) values for Ag-NP_{PVP}, AgNO₃ and K₂Cr₂O₇ were calculated using REGTOX-EV6.xls (Eric Vindimian <http://eric.vindimian.9online.fr/>), a curve fitting macro for Microsoft® Excel. Toxicity data for the algal and copepod tests were fitted to a sigmoidal curve and the Weibull (algal assays) and Hill (copepods assays) models were used to calculate Lethal Concentration (LC) and Effective Concentration (EC) values respectively. Statistical analyses were carried out using analyses of variance (ANOVA) followed by Dunnett's multiple comparison test. From these data, Lowest Observable Effects Concentrations (LOEC) and No Observable Effect Concentrations (NOEC) were calculated. These data analyses were performed using MINITAB® release 15 (MINITAB Inc. PA, USA). Statistical significance was accepted at $p \leq 0.05$.

3 Results

3.1. Nanoparticle characterisation

3.1.1. Dynamic Light scattering (DLS)

The average particle sizes of Ag-NP_{PVP} in all media and MilliQ water, as measured by DLS, are shown in Table 1. Ag-NP_{PVP} in MilliQ water showed hydrodynamic peaks at around 6 and 60 nm (Z-average value of 56.9 nm). In the case of the saline media, the values varied and there appeared to be no average particle size increase/decrease trend with increasing salinity. The CT10 ‰ media showed peaks at around 40 and 80 nm (Z-average value of 149 nm). The CT20 ‰ media showed two peaks in the size distribution of approximately 5 and 60 nm (Z-average value of 57.28 nm). The CT30 ‰ media showed two peaks in the size distribution at ca. 40 and 120 nm (Z-average value of 105.01 nm). The highest salinity media (NSW) showed two main size distribution peaks, at around 28 and 154 nm (Z-average value of 76.44 nm). All size determination results are summarized in Table 1. Although the polydispersity index was below 0.5 for all samples measured, indicating that Z-average is a suitable mean size to use to compare these samples, the presence of multiple peaks, indicates that a degree of aggregation/agglomeration occurs in the media and the Z-average as determined by DLS may not be the most appropriate technique for representing the size distribution of these particles in solution

For zeta potential measurements, it was only possible to obtain usable values in the MilliQ, CT10 ‰ and CT20 ‰ media samples, as the high conductivity of the samples with a salinity above 20 ‰ caused interference in the readings. The zeta potential of the particles measured in the MilliQ water and *C. tenuicorne* media at 10 ‰ and 20 ‰ were -39.17 mV, -8.68 mV and 10.32 mV respectively. The low zeta potential values are consistent with a tendency for the nanoparticles to agglomerate or aggregate, as indicated by the size distribution measurements.

3.1.2 Atomic Force Microscopy (AFM)

AFM images were captured and analysed for drop cast samples of Ag-NP_{PVP} from all media and MilliQ water. Figure 1 shows an example of an AFM image and corresponding topography of a cross section for Ag-NP_{PVP} from CT10‰. As seen from Figure 1, the silver nanoparticles form isolated aggregates when precipitated from the media. The cross section of the image shows one of these aggregates with a height greater than 150 nm and a base diameter of > 0.5 μ m. However, a significant number of features of dimensions ~10nm are also observed, indicating the presence of isolated nanoparticles. The average size and particle size distribution are described in Table 1.

3.1.3 Transmission Electron Microscopy (TEM) analysis

While both DLS and AFM indicate a significant degree of aggregation/agglomeration in the dispersions, TEM images indicate the presence of particles in the nanometer range (< 100 nm, ASTM International, 2006) in the stock solution (Farkas et al., 2010) and in the analysed exposure solutions 10000 μ g/L. All TEM measurements are described in Table 1. The presence of nanoparticles is clearly seen in the MilliQ water sample, in which approximately 48 % of particles were < 5nm, while, 30 % were between 5 – 10 nm in size. Of the exposure media, the smallest average size, based on TEM analysis, was for CT10 ‰ media with an average size of 16.0 ($\pm$ 5.8) nm. The average size for the CT20 ‰ media, CT30 ‰ media and the NSW (ca. 34 ‰) were 20.2 ($\pm$ 4.1), 19.3 ($\pm$ 6.3) and 18.9 ($\pm$ 5.2) nm respectively. As with the DLS analysis, there was no specific trend in particle size based on increasing/decreasing salinity. The Ag-NP_{PVP} in all saline media (except in CT30 ‰, where only dispersed particles were seen) were observed to form loose agglomerates at a concentration of 10000 μ g/L as well as being present as monodisperse particles in solution (Figure 2). In the calculated PSD (Figure 3), results indicated that there were no particles less

than 5 nm in diameter in any media type. The size distributions of the analysed particles were similar for all media, 7.9 – 33.7, 11.4 – 36.3, 9.7 – 39.6 and 9.9 – 34.9 nm respectively, for CT10 %, CT20 %, CT30 % and NSW. For all media types between 87 - 98 % of the measured particles were in the range 10 – 30 nm in diameter (Figure 3).

3.2 Ecotoxicity testing

3.2.1 *Tisbe battagliai*

The mean 48 hour EC₅₀ value for the reference chemical potassium dichromate with *T. battagliai* was 7.2 (95 % CI = 5.8 – 9.1) mg/L, which corresponds to previously published values for this species and chemical (Macken et al., 2008). No deleterious effects of PVP were observed, therefore, toxicity due to the PVP capping agent in these experiments can be ruled out.

Toxicity data for Ag-NP_{PVP} and AgNO₃ to *T. battagliai* are shown in Figure 4a and b and Table 2. The LC₁₀ and LC₅₀ values indicate that the toxicity of Ag-NP_{PVP} was statistically greater than the AgNO₃ toxicity. Despite the calculation of these values it can be seen from Figure 4 that the silver salt had a greater toxic effect at higher concentrations, while the silver ENP appeared to be more toxic at lower concentrations (see Figure 4c for a graphical comparison of the mortality data). For all experiments, the test vessels were treated with solutions of suitable concentrations overnight prior to test initiation and rinsed with MilliQ water. This was done so as to remove any potential binding sites for the silver test solutions on the plastic test vessel surfaces. The control mortality and physico-chemical parameters measured at the start and end of all experiments were within the recommended limits (ISO, 1999).

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340 3.2.2 *Ceramium tenuicorne*

341 The mean 7 day EC₅₀ value for the K₂Cr₂O₇ with *C. tenuicorne* was 3.9 mg/L (95 %
342 CI = 2.6 - 5.7), which was consistent with previous studies with this species (Macken,
343 unpublished) and no toxicity was observed with the PVP capping agent. The mean increase in
344 length over a 7 day test period, of six algal pieces, in the three proposed test salinities was
345 measured. Growth rate was observed to increase with increasing salinity, 1.12, 1.46 and 1.62
346 respectively for 10, 20 and 30 ‰ media. All salinities allowed for good growth of the algae,
347 sufficient to meet all validity criteria for standard guideline testing (ISO, 2010), however, for
348 continuous cultures, it was observed that the 20 ‰ cultures were the best performing for
349 long-term culture maintenance.

350 The control growth rates met with the validity criteria in all experiments (increase in
351 length over seven days by a factor of greater than 3, compared to the starting length [ISO,
352 2010]), therefore implying that the use of parafilm and reduced sample volume had no
353 adverse effects on the test system. Table 2 summarises the ecotoxicity data for all test
354 salinities with both Ag-NP_{PVP} and AgNO₃. As was observed with the *T. battalgiai* assays, the
355 toxicity of the silver ENPs was greater than that of AgNO₃ (Table 2 and Figure 5) to the test
356 organism when looking at EC values. Despite this, when NOEC/LOEC values were
357 calculated, the difference in toxicity changed and the ENPs can only be considered more
358 toxic at the highest salinity (CT30 ‰). The toxicity of the Ag-NP_{PVP} increased with
359 increasing salinity (EC₅₀ values of 1872.5 (95 % CI = 954.5 – 3757.6), 393.5 (95 % CI = 954.5 –
360 3757.6) and 26.7 (95 % CI = 12.6 – 69.8) µg/L, at CT10 ‰, CT20 ‰ and CT30 ‰,
361 respectively). The observed toxicity of AgNO₃ to *C. tenuicorne* at all salinities was similar,

with a slight, but not statistically significant, increase in toxicity between 10 ‰ and 20 ‰ (based on EC values).

4 Discussion

The toxicity of both the ionic silver and Ag-NP_{PVP} assayed in this study were high compared to other similar studies in the literature. No comparable toxicity data for *T. battagliai* and silver or silver ENPs are available, however, there is some information on other marine copepods. Pedroso et al. (2007) found that at 30 ‰ the 48 h EC₅₀ for *Acartia tonsa* was 173 µg/L (dissolved silver) indicating the harpacticoid, *T. battagliai* was more sensitive to silver toxicity (48 h EC₅₀ at 34 ‰ = 90.9 µg/L). In relation to species specific differences, there are limits to identifying the most sensitive of the two species assayed in this study. Different salinities, exposure periods and endpoints (e.g. lethality and growth) are being assessed. However, it is clear that the sensitivity of the two species differs depending on these factors and it is important to incorporate a battery of endpoints and exposure periods for ERA in order to allow for the protection of the marine and estuarine environment.

The higher toxicity of silver ENPs compared to ionic silver, with *T. battagliai* may be indicative of a potential particle specific mode of action on the toxicity and has been observed by other authors (Griffitt et al., 2008; Navarro et al., 2008). The increase in toxicity could be as a result of increased surface area and therefore increased capacity to release ionic silver from the surface of the particle.

C. tenuicorne was also observed to be slightly more sensitive to Ag-NP_{PVP} than ionic silver, however, differences in sensitivities were not as evident as those observed with

T. battagliai at all salinities. After exposure in algal media of 10 ‰, the EC₁₀/EC₅₀ values were similar for both Ag-NP_{PVP} and AgNO₃. The difference in sensitivity between the two test chemicals became much more pronounced with increasing exposure media salinity. These differences were particularly pronounced at 30 ‰. There did not appear to be a marked increase in toxicity of AgNO₃ with increasing salinity. In contrast, there was a consistent increase in toxicity up the salinity gradient with Ag-NP_{PVP}. The toxicity of metals in aquatic systems is influenced by a variety of factors such as organic matter content (Nadella et al., 2009) and salinity (Verslycke et al., 2003). These results may indicate that an increase of chloride ions or other inorganic ligands within the media are changing the behaviour and bioavailability of the silver ENPs to the test species. Previously published work, investigating the effects of salinity on the toxicity of contaminants in the marine environment to invertebrates, have observed an increase in toxicity with decreasing salinity (Verslycke et al., 2003; Kwok and Leung, 2005; Pedroso et al. 2007). Ytreberg et al. (2011) investigated the effects of salinity and organic matter on the toxicity of Cu to *C. tenuicorne* and concluded that the effects of salinity on Cu toxicity were not clear, as both a positive and negative effect was observed, a decrease in toxicity with increasing salinity in the presence of organic matter but no decrease in the absence of organic matter.

As in this work, an increase in toxicity with increasing ionic strength has also been observed in the literature (Erikson et al., 1998). This is the opposite of what would be expected if only the free silver ions were responsible for the toxic response. Due to the high ionic strength of seawater and the prevalence of inorganic ligands such as chlorine, silver complexation will not be dominated by NOM, as in freshwater, but instead speciation is dominated by the formation of strong chloro-complexes (Cowan et al., 1985; Cowan et al., 1993; Reinfelder and Chang, 1999). Without definite information on the speciation and complexation of silver in the media and seawater in our study, it is not possible to make clear

conclusions on the causes of the observed toxicity or the effects of changing ionic strength. Unfortunately due to the low levels of silver, limitations in analytical capabilities and small volumes of test material, speciation could not be investigated in greater detail in this particular study.

As well as understanding the speciation of silver within seawater, an understanding of the chemical nature of the exposure medium is also important in determining the behaviour and bioavailability of silver and silver nanoparticles. In this study all of the exposure solutions were prepared from NSW, however, the algal media was also enriched with a variety of trace metals, salts and vitamins, including nitrogen, phosphorus, iron, and trace amounts of cobalt, copper, zinc and manganese (ISO, 2010). One important factor for consideration is the presence of chelating agents within exposure media (e.g. EDTA). In this work the ISO *C. tenuicorne* media contained low levels of EDTA (final concentration of approximately 300 µg/L). Despite this, it is unlikely that the presence of EDTA can completely explain the difference in sensitivity between the two species, as EDTA has been shown to have very a low affinity for silver (Zuiderveen and Birge, 1995).

In order to further try and explain the results of our study we have considered the potential physiological susceptibility of the algae itself under differing environmental regimes. Russell (1985) conducted an investigation of macroalgae in the Baltic Sea and found *C. tenuicorne* was the least likely to cope with any large increase in salinity. Despite, the algae used in our study going through an adaptation period of several weeks, this may help to explain the increased sensitivity of this species with increasing salinity. In order to respond to external salinity changes this particular seaweed, has to alter its water content and the concentration of inorganic ions (Lobban and Harrison, 1994; Ferguson and Hogstrand, 1998). Due to the fact that the Ag-NP_{PVP} toxicity increased with increasing salinity and the AgNO₃

toxicity did not increase, there could also be some sort of ENP effect that is enhanced in the presence of increasing Cl^- .

It is believed that no single technique for ENP characterisation is without its artefacts or can be employed in all cases for nanoparticle characterisation (Domingos et al., 2009). Therefore, in order to try and counteract this problem a battery of characterisation techniques were employed in this study. The dispersion system employed in this study was intended to reflect the realistic conditions and behaviour of the ENPs on entry into the marine environment and so no manipulations to force the ENPs into solution were conducted (e.g. use of solvents) (Kato et al., 2009).

In recent years, TEM has allowed scientists to obtain images of individual nanoparticles. However, during this analysis only a small fraction of the total sample is characterised. Therefore, the results may not be representative of the total sample being assayed and it is easy to acquire incorrect information on exposure solutions (Nowack and Bucheli, 2007). There are several potential artefacts associated with TEM and ENP characterisation, e.g. meniscus-based artefacts and co-precipitation with inorganic salts (Domingos et al., 2009). In saline media, the same effect could cause interference with the visual analysis of the sample, as any impurities, or salt in the sample, could make it difficult to distinguish the ENPs from these contributions on the air-dried grid.

Compared to the TEM results, the results of DLS measurements of the average particle size of Ag-NP_{PVP} in all media types, were far greater. Similar results have been observed by other authors (Domingos et al., 2009; Farkas et al., 2010; Hassellöv et al., 2008). It is important to note, that in DLS the diameter being measured is the hydrodynamic diameter, which is not only dependent on the core of the ENP but also on any surface

structures (i.e. PVP capping agent) and the concentration of ions in the medium. Therefore, the particle size can be larger than that measured by electron microscopy techniques.

In this study, some of the raw data from the DLS measurements did not meet the quality criteria (data generally showing high polydispersity). This polydispersity may have been due to the fact that, with the addition of the silver ENPs to seawater, silver chloride precipitated out of solution and caused the ENPs to sediment out. Immediately following the addition of ENPs there was no visible sedimentation of particles in the test exposure solutions, however, after microscopical examination (light microscope) at test termination there was apparent fall out of particles on the bottom of the test wells in all experiments. For zeta-potential measurement salinities above 20 ‰ were unable to provide usable, reproducible values. The high conductivity of the saline samples was most likely the cause of the interference in the readings. In the *C. tenuicorne* media there was a shift in zeta-potential from a negative (CT10 ‰ = -8.68) to a positive (CT20 ‰ = 10.32) value with increasing salinity. This shift of zeta-potential towards a positive value at higher salinities suggests that a possible charge repulsion mechanism is becoming less dominant (Oo and Ong, 2010). It has also been shown that inorganic ions (e.g. Cl⁻) in solution can interact with charged surfaces by non-specific or specific ion adsorption, which will affect the isoelectric point. In some situations, as shown in this study, specific ion adsorption can lead to charge reversal of the surface (Zeta Sizer Nanoseries, 2004).

In order to conduct a more thorough risk assessment of nanoparticle exposure it is important to have some specific information. Within the environment their quantity, persistence and dispersion is of vital importance. Without this information it is impossible to conduct a comprehensive risk assessment and herein lies the problem. As previously mentioned, measured total/bulk silver levels in the marine and estuarine environment are very low, 0.04 – 31 ng/L (Kramer et al., 2002) and it is, at present, impossible to distinguish bulk

silver released into the marine environment from silver ENPs present. Therefore, we have no true understanding of what the realistic levels of these materials in the aquatic environment are. It is also likely that the exposure levels used in the laboratory are far higher than the natural levels in the environment (Ward et al., 2006). Potentially, these high exposure concentrations, used to cause a toxic effect in some laboratory experiments, may result in different uptake mechanisms and rates than those that would occur at environmentally realistic levels of silver or ENP silver (Fortin and Campbell, 2000). Therefore, making the results of laboratory based ENP studies difficult to incorporate into nanoparticle risk assessment at present, however, studies on the sensitivity and effects of species exposed to ENPs can help provide information that may, in time, lead to a better understanding and more appropriate use of toxicity information for ERA of these materials. In addition, the guidelines need to be clarified to incorporate definitive guidance on the characterisation of ENPs in environmental media. Guidance on the preparation of ENP solutions for ecotoxicological assessment needs to be harmonised in order to yield comparable interlaboratory results for future ERA. The development of additional nano-specific guidance documents are at present being discussed but it remains a complicated and developing area of regulatory ecotoxicology.

5 Conclusion

Silver ENPs have the potential to cause toxic effects to marine macrophytes and invertebrates. Given the sensitivity of the two species used in this study, and their widespread distribution in the marine environment, their susceptibility is of utmost importance. In developing methodologies for the assessment of ENP toxicity, it is vital that the most suitable species and endpoints be identified. At present definitive guidelines on the assessment of the risks associated with ENP release and effects in the environment do not exist. Considering

that increased salinity was observed to have such a marked effect on the toxicity of the ENPs employed in this study, it may not be appropriate to use freshwater data to extrapolate risk within the marine environment. In addition, as the ISO methods stand, they may not be completely suitable for the assessment of ENPs. Despite the test design, organism's sensitivity and environmental relevance, the methods lack specific information on sample preparation, dosing and quantification of exposure concentrations. Although the characterisation techniques employed were unable to correlate ENP behaviour and salinity it is apparent that some process is causing an increase in toxicity with increasing salinity. As similar increases in toxicity of AgNO₃ with increasing salinity were not observed, it can be concluded that it is not merely the chemical nature of the ENPs responsible for the effect. Therefore, the observed increase in toxicity is most likely due to a combination of the surface properties of the ENPs and their reactivity within highly saline media, and the behaviour of the silver ions and complexation. There is a definite need for a more thorough understanding of the modifying effects of changing environmental parameters. This knowledge, along with suitable guidelines, is required in order to contribute more accurately to an ERA of ENPs introduced into the marine environment.

Acknowledgements

The authors sincerely wish to thank Dr. Britta Eklund at the University of Stockholm for providing a starting culture of *C. tenuicorne*, Dr. Norbert Roos at the University of Oslo, for helping with the TEM imaging and Dr. Luke O'Neill for his time and efforts with the AFM imaging. This work was conducted under the framework of the INSPIRE programme, funded by the Irish Government's Programme for Research in Third Level Institutions, Cycle 4,

National Development Plan 2007-2013, supported by the European Union Structural Fund.
and The Norwegian Research Council under the Yggdrasil mobility programme.

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770

771 **Figures**

772

773

774

775 **Figure 1** An example of AFM image of Ag-NP_{PVP} in CT10 ‰ with underlying topographic
776 profile.

777

Figure 2 TEM images (scalebar: 200 nm) of Ag-NP_{PVP} in NSW (a - b), CT10 % (c - d), CT20 % (e), CT30 % (g - h) and the corresponding average size in all media (f).

Figure 3 Particle size distribution (PSD) from TEM image analysis of Ag-NP_{PVP} in different media (n = 100 for measurements in all media).

787 (a)

788

789 (b)

790

791

792

793 **Figure 4** Toxicity of (a) Ag-NP_{PVP} (b) AgNO₃ to *Tisbe battagliai* exposed for ■ 24 hours and
794 □ 48 hours.* indicates statistical significance from the control ($p \leq 0.05$). Data are mean $\pm$ SD
795 (n = 3). CV for the controls ranged from 0.00 – 2.94 %.

796

797 (a)

798

799 (b)

800

801

802 **Figure 5** Growth inhibition of *Ceramium tenuicorne* after 7 days exposure to Ag-NP_{PVP} (a)
803 and AgNO₃ (b) at 10 ‰ (□), 20 ‰ (■) and 30 ‰ (▣). * indicates statistical significance from
804 the control ($p \leq 0.05$). Data are mean $\pm$ SD (n = 3). CV for the control ranged from 0.15 –
805 1.33 % (10 ‰), 5.29 – 13.07 % (20 ‰) and 15.7 – 16.65 % (30 ‰)
806

807 **Table 1** Particle size analysis and characterisation results for Ag-NP_{PVP} in all test media.

Media type	Method	Concentration (mg/L)	Mean PS (nm)	SD (nm)	Min (nm)	Max (nm)
MilliQ water	DLS	0.1	56.9	7.8	-	-
	TEM	10	7.0*	-	1.0*	60.0*
	AFM	10	18.5	6.2	5.9	38.2
10 ‰ 2A Media	DLS	0.1	149.0	29.8	-	-
	TEM	10	16.0	5.8	7.9	33.7
	AFM	10	24.7	38.7	6.5	280.1
20 ‰ 2A Media	DLS	0.1	57.3	3.2	-	-
	TEM	10	20.2	4.1	11.4	36.3
	AFM	10	13.8	7.1	5.5	56.4
30 ‰ 2A Media	DLS	0.1	105.0	18.5	-	-
	TEM	10	19.3	6.3	9.6	39.6
	AFM	10	24.0	23.7	5.7	120.3
NSW	DLS	0.1	76.4	39.2	-	-
	TEM	10	18.9	5.2	9.9	34.9
	AFM	10	12.2	7.8	55.4	71.7

808 SD = Standard deviation, NSW = Natural Seawater, DLS = Dynamic Light Scattering, TEM = Transmission Electron Microscopy, AFM =
809 Atomic Force Microscopy

810 * Approximately 48 % of particles were < 5nm, while, 30 % were between 5 – 10 nm in size (Farkas et al., 2010)

811

812 **Table 2** Ecotoxicity data for *Tisbe battagliai* and *Ceramium tenuicorne* assayed with silver nanoparticles (Ag-NP_{PVP}) and silver nitrate (AgNO₃).

Test Substance	Test Species	Salinity (‰)	Duration	EC ₁₀ /LC ₁₀ ^a (µg/L) ^a	EC ₅₀ /LC ₅₀ ^b (µg/L) ^b	NOEC ^c /LOEC ^d (µg/L)
Ag-NP _{PVP}	<i>T. battagliai</i>	34	24 h	3.8 (2.4 – 5.6)	30.8 (26.2 – 35.7)	3.2/5.6
		34	48 h	0.9 (0.6 – 1.7)	7.9 (6.2 – 10.3)	3.2/5.6
Ag-NP _{PVP}	<i>C. tenuicorne</i>	10	7 d	32.5 (4.0 – 129.6)	2120.4 (1199.9 – 3316.8)	100/500
Ag-NP _{PVP}	<i>C. tenuicorne</i>	20	7 d	1.5 (0.1 – 11.0)	373.9 (179.8 – 782.7)	50/100
Ag-NP _{PVP}	<i>C. tenuicorne</i>	30	7 d	0.1 (0.007 – 1.4)	26.7 (12.6 – 69.8)	5/10
AgNO ₃	<i>T. battagliai</i>	34	24 h	12.7 (0.8 – 258.7)	167.3 (52.8 – 481.8)	32/56
		34	48 h	8.7 (2.0 – 102.5)	90.9 (38.0 – 238.9)	10/32
AgNO ₃	<i>C. tenuicorne</i>	10	7 d	42.2 (6.4 – 143.8)	3606.5 (2318.2 – 5378.6)	10/50
AgNO ₃	<i>C. tenuicorne</i>	20	7 d	10.9 (0.6 – 81.4)	2246.6 (1020.8 – 4056.4)	50/100
AgNO ₃	<i>C. tenuicorne</i>	30	7 d	11.4 (0.4 – 105.6)	2312.2 (1038.4 – 4819.6)	500/1000

813 ^aEC₁₀/LC₁₀ values and corresponding 95 % confidence intervals

814 ^bEC₅₀/LC₅₀ values and corresponding 95 % confidence intervals

815 ^cNOEC, no observed effect concentration, the highest observed concentration at which no significant effect ($p \leq 0.05$) was detected

816 ^dLOEC, lowest observed effect concentration, the lowest concentration of the tested concentration at which a significant ($p \leq 0.05$) effect was detected

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