How are macroinvertebrates of slow flowing lotic systems directly affected by suspended and deposited sediments?

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Sediment deposition may be more directly detrimental to macroinvertebrates of lowland rivers than suspended sediments.

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ABSTRACT

The effects of suspended and deposited sediments on the macroinvertebrates are well documented in upland streams but not in slower flowing lowland rivers. Using species found in lowland lotic environments, we experimentally evaluate mechanisms for sediments to affect macroinvertebrates, and in one experiment whether salinity alters the effect of suspended sediments. Suspended kaolin clay reduced feeding of *Ischnura heterosticta* (Odonata: Coenagrionidae) at high turbidity (1000–1500 NTU) but had no effects on feeding of *Hemianax papuensis* (Odonata: Aeshnidae) and *Micronecta australiensis* (Hemiptera: Corixidae). In freshwater (0.1 mS/cm), survival of *Ischnura aurora* was poor in clear water, but improved with suspended kaolin. Growth and feeding of *I. aurora* were unaffected by suspended sediments and salinity. Burial (1–5 mm) of eggs with kaolin or sand reduced hatching in *Physa acuta* (Gastropoda: Physidae), *Gyraulus tasmanica* (Gastropoda: Planorbidae) and *Chironomus cloacalis* (Diptera: Chironomidae). Settling sediments may pose greater risk to lowland lotic invertebrates than suspended sediments.

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1. Introduction

Sediments affect a greater length of rivers than any other pollutant in the USA (Parkhill and Gulliver, 2002). In Australia, along with salinity and nutrients, sediments are regarded as one of the top three most important river contaminants (Lovett et al., 2007). Sediments are generated by natural erosion and many anthropogenic activities including agriculture, grazing, forestry, gravel roads, mining and construction. Sedimentation of fast flowing streams with riffle habitats has adverse effects on freshwater (otherwise benthic) macroinvertebrates by increasing downstream drift (e.g. Doeg and Milladge, 1991), reducing the ability of drifting invertebrates to re-attach to the stream bed (see Bilotta and Brazier, 2008), decreasing the feeding efficiency of filter feeders and algal grazers and by burying habitat and the in-filling of spaces between larger substrate, i.e. boulders, cobbles and pebbles (Waters, 1995). Salmonid fish are sensitive to sedimentation by the in-filling of space in gravel spawning grounds and the clogging and abrasion of their gills, leading to decreased immunity to disease and osmotic dysfunction (Bilotta and Brazier, 2008). Cyprinid fish are, however, much more tolerant of sediments.

When considering the impacts of sedimentation on lotic ecosystems, it is important to differentiate between fast and slow flowing streams as each are likely to experience different modes of impact. Fast flowing streams and riffle habitat are rare in much of Australia and in other areas where low relief dominate. Sediment-sensitive salmonid fish are native only to the mid to high northern latitudes (but widely introduced elsewhere). Relative to streams in higher relief environments, lowland streams in Australia often have high suspended sediment concentrations even during low flow and this may be natural. In slow flow, passive drift of macroinvertebrates is likely to transport benthic invertebrates shorter distances than in faster flowing rivers. The effect of increased drift from sedimentation may thus be less important than in fast flowing streams. Additionally the beds of these streams are often dominated by fine particles (sand, silt and clay) and the burial or in-filling of spaces by sediment is thus unlikely to play an important role in any effects of sediment on aquatic biota. Much of the effect data generated for organisms from regions dominated by fast flowing stream, may not be relevant in the slow flowing rivers. There have been fewer studies of the effects of settling sediments and suspended sediments on invertebrates of these streams.

The Australian and New Zealand water quality guidelines recognise that sediment may have differing effects in lowland and upland rivers (and also lake/reservoirs) by splitting trigger values

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between these environments (ANZECC, 2000). Based mostly on typical turbidity values in localities regarded as minimally disturbed, the trigger values for lowland rivers are: south-east Australia of 6–50 NTU, tropical Australia of 2–15 NTU, south-west Australia 10–20 NTU, south-central Australia 1–50 NTU and New Zealand 5.6 NTU. The extent of impact from suspended sediments levels above these turbidity ranges in these environments is uncertain. In western Queensland, for example, a number of rivers are regarded as having high levels of suspended sediments (e.g. up to 1800 mg/L, Negus and Farthing, 2006) but based on their macroinvertebrate community they are regarded as being healthy (Prior, 2006; Steward, 2006). However, as slow flowing turbid streams usually have substantial agriculture, grazing or other human activities within their catchments, it is generally not possible to identify reference sites to provide benchmarks for healthy macroinvertebrate communities. Additionally, while restricted to the edges, turbid inland rivers still have high primary production that support invertebrate and fish communities (Fellows et al., 2008). So it is uncertain whether the macroinvertebrate communities of turbid streams in western Queensland, and elsewhere, are affected by sediments.

One approach to determine whether sediments directly affect invertebrates of slow flowing rivers is to experimentally determine the relevance of hypothesised mechanisms by which sediments might affect invertebrates. This paper experimentally determines whether selected macroinvertebrates, typically found in lowland rivers are affected by suspended and deposited sediments. Three hypothesised mechanisms of direct effects of sediments are investigated: reduced feeding efficiency of visual predators; chronic effects of suspended sediment requiring greater expenditure of energy resulting in less energy for growth; and the survival of eggs buried by settling sediment. We only consider whole organism responses or endpoints (e.g. feeding rates, survival, growth), as they are likely to be more ecologically relevant than sub-organism biomarkers (e.g. physiological and biochemical responses) (see Forbes et al., 2006). We do not consider here indirect effects (i.e. effects delivered via ecological interactions).

2. Materials and methods

2.1. General methods across all experiments

The dilutant water and that used as a control in all experiments was carbon and sand filtered (de-chlorinated) Melbourne tap water, hereafter referred to as wet lab water (WLW), which has a salinity, as electrical conductivity (EC), of approximately 0.1 mS/cm and turbidity < 1 NTU. As standardised suspended sediments, we used kaolin (clay) which consists mostly (85–95%) of the mineral kaolinite $Al_2Si_2O_5(OH)_4$. Kaolin was chosen as it is a fine clay and has low toxicity to aquatic species (WHO, 2005), thus any observed effect would be from the suspended sediments and not from toxicity of specific mineral(s) or attached pollutants.

While we present results in terms of turbidity (NTU), the relationship between turbidity and total suspended sediments (TSS in mg/L), with kaolin and WLW used in our experiments is:

$$TSS = 0.601 \times Turbidity \tag{1}$$

The standard error on the constant is 0.0024, and $r^2 = 0.99995$.

2.2. Effects of suspended sediment on feeding efficiency of visual invertebrate predators

The effect of suspended sediment on feeding efficiency of visual macroinvertebrate predators was tested using three invertebrate species. The damselfly larva *Ischnura heterosticta* (Odonata: Coenagrionidae) collected from either Mt. Emu Creek at Western Hwy or the Barwon River at Pollocksford Bridge, the dragonfly larva *Hemianax papuensis* (Odonata: Aeshnidae) from Newport Lakes, Newport and the adult water bug *Micronecta australiensis* (Hemiptera: Corixidae) from Newport Lakes with all collection locations being in Victoria, Australia. These species were chosen because they are predators with large eyes and thus likely to be visual predators, are widely and regularly found in slow flowing rivers and were obtainable in numbers sufficient for the experiments.

Similarly sized individuals of each species were selected and any individual that had recently (<3 d) moulted was excluded. To ensure that all individuals of a species were equally hungry, prior to the experiment all were given equal feeding opportunity. During

this acclimation and during the experiment replicate individuals of each species were held in 60 mL of water. The acclimation consisted of feeding *ad lib* for 2 days with brine shrimp *Artemia salina* and then no feeding on one day, both in WLW. We do not know the diet of the predators in nature, but all readily ate *A. salina* and previously *I. heterosticta* preferentially consumed them over the claderocern *Daphnia caranata* (Kefford et al., 2006). Then for each species, replicate individuals were placed individually in treatments with 20 *A. salina* for 1 h. Five replicate *H. papuensis* were used per treatment and 9 replicates of the other two species. The treatments were a control of WLW (<1 NTU), and WLW with various concentrations of kaolin added. The number of treatments used for each species depended on the number of individuals available and the specific treatments were determined based on results of previous experiments. After exposure to treatments of kaolin, test organisms were removed and the number of uneaten *A. salina* counted. The predators were placed individually in a second identical container but with 60 mL of (clear) WLW (regardless of the treatment they had just been in) for another hour. At the conclusion of the second hour, 20 (new) *A. salina* were placed with each predator (in the WLW) for another hour, after which the predators were removed and the number of uneaten *A. salina* counted. The purpose of the second (post exposure) feeding in (clear) WLW was to establish a reference level to account for any inter-individual variation in hunger, prey catching ability, etc. If high turbidity reduced feeding efficiency, it would be expected that in high turbidity treatments individuals would catch relatively less *A. salina* but when given a second feeding opportunity in (clear) WLW there should be no differences in the amount eaten between individuals.

For each individual and treatment, the proportion of *A. salina* consumed was calculated for the first feeding opportunity and is hereafter referred to as the proportion eaten in the treatment. The proportion eaten in the second feeding opportunity, hereafter proportion eaten in WLW, was also calculated. Finally the ratio of these two proportions was calculated, referred to as the ratio of proportion eaten in treatment: WLW. If turbidity affects the feeding efficiency, then this ratio should decrease with increasing turbidity.

A second experiment was run with *I. heterosticta* that was identical to the above but with different levels of turbidity (using the results of the first experiment as a range finder) and with 10 replicate *I. heterosticta* per treatment.

Differences between treatments were examined using a single factor ANOVA (turbidity) or if the ANOVA's assumption of homogeneity of variance and normal distribution could not be met, a non-parametric Kruskal Wallis Test was used.

2.3. Effects of suspended sediment exposure on feeding efficiency by mechanisms other than reduced vision

In the previous experiment, individuals were exposed to suspended sediments for only 1 h. It is possible that longer exposure to suspended sediments might have effects on feeding rates by mechanisms other than reduced vision. For examples, individuals exposed to suspended sediments may spend more energy moving their gills and other fine structures to prevent sediment from settling. To establish whether the effect of suspended sediment on feeding efficiency of visual invertebrate predators was the result of reduced vision and not any other mechanism a second experiment was run using *I. heterosticta*. In this experiment, *I. heterosticta* were conditioned, as above, so that they were equally hungry. Then 9 replicates of individually housed *I. heterosticta* were placed into various turbidity treatments for 40 h. After which they were placed into individual 50 mL containers of (clear) WLW for 1 h. After that 30 *A. salina* were added, and after 1 h the *I. heterosticta* were removed and the number of uneaten prey counted. If suspended kaolin affected the hunger or prey catching ability of *I. heterosticta* by mechanisms other than reduced vision, then there should be differences in the proportion of prey caught between the treatments.

2.4. The lethal and sub-lethal chronic effects of suspended sediments on Ischnura aurora

Suspended sediments may damage the gills, small appendages and other fine structures of invertebrates (Langer, 1980, in Bilotta and Brazier, 2008). Additionally to reduce sediment build-up on these fine structures, individuals may need to spend energy moving them. For such effects to be of ecological relevance they would need to affect whole organism responses. Consequently we measured growth rates (mm/d), feeding and survival at two different turbidity treatments (100 and 1000 NTU) and a control (≈ 0.1 NTU). The kaolin was kept in suspension by aeration applied equally to all treatments. We originally planned to conduct this experiment with *I. heterosticta* but they were unavailable, so we used the related species *I. aurora*. A previous study showed that growth of *I. heterosticta* was higher over the EC range 5–20 mS/cm than 0.1–1 mS/cm (Kefford et al., 2006). Here we examined whether *I. aurora* was similarly affected by EC and whether EC modified the effect of suspended sediments. Consequently we also had a second factor: EC (0.1 and 10 mS/cm @ 25 °C). Based on our experience with *I. heterosticta* and that *I. aurora* was collected from a relatively saline site (6–7 mS/cm), we expected *I. aurora* to grow faster and possibly have lower mortality at 10 mS/cm than 0.1 mS/cm.

The experimental methods were essentially identical to those used with *I. heterosticta* and are presented in Kefford et al. (2006) and are only briefly described here. *I. aurora* were collected from Newport Lake (EC 6.1 mS/cm and 16.1 NTU). Prior to exposure, the body length (i.e. not including gills) and (maximum) head width (mm) of each individual was measured with an eyepiece micrometer. Six replicate *I. aurora* were