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Chironomid assemblage structure and morphological response to pollution in an effluent-impacted river, Eastern Cape, South Africa

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ABSTRACT

Achieving sustainable freshwater resource management and conserving biodiversity require accurate and reliable pollution impact assessment. Biomonitoring is an integrative and holistic approach to assessing impact of pollution using biological indicators. In this study, we use the assemblage structure of the freshwater macroinvertebrate family Chironomidae to assess the impact of pollution in a South African river: the Swartkops River. Based on morphological deformities in the mentum of selected genera, we developed a new index, the extended toxic score index (ETSI), to assess the extent of deviation of ecological conditions at the impacted sites from that at the control site. Four sites i.e. Sites 1–4 characterised by different degrees of pollution were selected for chironomids and physico-chemical sampling over a period of three years (August 2009–September 2012). The ETSI was developed by taking into account the severity of deformities, and genera tolerances to pollution. The index was based on the deformities in the mentum of the five most dominant genera: *Chironomus*, *Polypedilum*, *Tanytarsus*, *Dicrotendipes* and *Cricotopus*. The values of nutrients, electrical conductivity and turbidity were elevated, and dissolved oxygen depleted at Sites 2, 3 and 4 compared with Site 1. Metal concentrations in the water column were relatively low (<0.3 mg/l). Site 1, i.e. the control site, supported more chironomid species (30 species). Site 2, situated upstream of the Kelvin Jones wastewater effluent discharge point, but influenced by diffuse pollution sources, had more species (22 species) than Sites 3 (10 species) and 4 (18 species). The species assemblage structure clustered largely by sites than by seasons, with the assemblage at Site 1 distinct to those at Sites 2, 3 and 4. The newly developed ETSI was sensitive to pollution, distinguishing between the ecological conditions at the sampling sites. The index placed Site 2 in Category D, indicative of poor ecological condition and Sites 3 and 4 in Category E, indicative of very poor conditions. We found the ETSI to be an appropriate and useful deformity-based tool for biomonitoring, and it can easily be modified and used in other river systems, both locally and internationally.

1. Introduction

Freshwater pollution is threatening the functionality and sustainability of aquatic ecosystems and their services (UN-Water, 2011, 2012). The impact of freshwater pollution is traditionally evaluated by measuring the values of water chemistry variables (Palmer et al., 2004). Time series forecasting models, including seasonal autoregressive integrated moving average (SARIMA), autoregressive moving average (ARMA), autoregressive integrated moving average (ARIMA) and autoregressive artificial neural network models, have been developed for pollution management by predicting and forecasting seasonal, monthly and annual run-off, discharges and flows in rivers and reservoirs (Pekarova et al., 2009; Faruk, 2010; Valipour et al., 2012, 2013; Valipour, 2015). These models are important in terms of forecasting and predicting the pollution dilution capacity of rivers and reservoirs. Although physico-chemically based pollution management is important, the analytical expense and expertise needed, and the vast amount of chemicals dissolved and suspended in water, make exhaustive physico-chemical analyses uneconomical and practically impossible. Furthermore, the effects of pollutants and their by-products on the biotic component of the aquatic ecosystem cannot be fully assessed using physico-chemical analysis alone (Palmer et al., 2004). Biomonitoring on the other hand, provides a holistic measure of aquatic ecosystem health by integrating the multiple and cumulative effects of chemical, physical and biological stressors

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(Rosenberg and Resh, 1993; Bonada et al., 2006). Biomonitoring also provides information needed for biodiversity conservation; e.g. the Dragonfly Biotic Index (DBI) (Simaika and Samways, 2009).

Macroinvertebrate-based biomonitoring undertaken by identifying taxa to families may not provide detailed information on both taxonomic and functional diversity, as several groups of species, including rare, endemic, endangered and pollution tolerant/sensitive species, would have been aggregated into one family; limiting the interpretive power of such data. Furthermore, since systematics is not based on functional characteristics, species and even genera within the same family may have different ecological requirements and are likely to respond differently to pollution (Lenat and Resh, 2001; Schmidt-Kloiber and Nijboer, 2004). For example, despite the high functional and taxonomic diversity of the family Chironomidae (Armitage et al., 1995; Ferrington, 2008), it is often considered pollution tolerant. Chironomids are among the most diverse groups of aquatic macroinvertebrates, exhibiting a wide-range of tolerance/sensitivity to pollution, and play an important role in the aquatic food web (Armitage et al., 1995). For these reasons, the bioindicator potential of the chironomids is worth exploring in a region with paucity of information.

The need to assess sub-lethal biotic effects of pollutants has led to the use of morphological deformities in freshwater biomonitoring (Odume et al., 2012a). Morphological deformities in chironomids are considered both as an early warning signal of aquatic pollution and as an endpoint related to ecological traits such as feeding and development (Janssens de Bisthoven et al., 1998). Incidences of deformities greater than 8% in chironomid larvae are usually taken as an indication of toxic in-stream stress (Nazarova et al., 2004; Ochieng et al., 2008). However, ecological conditions differ between aquatic ecosystems and between regions, thus applying such an arbitrary reference deformity level for all studies becomes questionable. Furthermore, since different chironomid species are differentially sensitive to pollution, 8% incidences of deformities in different chironomid assemblages consisting of species of different sensitivity to pollution would not necessarily indicate the same level of stress.

Lenat (1993) developed the toxic score index (TSI) for assessing impact of pollution based on the mentum of *Chironomus* spp. Other authors e.g. Al-Shami et al. (2011) applied the index for freshwater biomonitoring. The TSI is based on classes of deformities, and each class is assigned a weighting factor according to its apparent severity. Class I: larvae having slight deformities, which are difficult to distinguished from breakage or abrasion; Class II: larvae having severe and clear deformities including extra teeth, fused teeth, missing teeth and distinct asymmetry; Class III: larvae having at least any of two Class II types of deformity (Lenat, 1993). Weighting factors 1, 2 and 3 are assigned to Classes I, II and III, respectively. The TSI is expressed as:

$$\mathrm{TSI} = \frac{[\text{No. of Class I} + 2(\text{No. of Class II}) + 3(\text{No. of Class III})] \times 100}{\text{Total No. of larvae}}$$

The TSI has a number of limitations. First, it is based only on *Chironomus* spp., thereby ignoring deformities occurring in other genera and making sites comparison difficult if *Chironomus* does not occur in sufficient number (at least 25 larvae) across all sampling sites. Second, it does not account for differential sensitivity among chironomid species to pollution (Odume and Muller, 2011). Third, the definition of the Class I deformities is ambiguous because they are difficult to distinguish from breakage (Hämäläinen, 1999).

The identified limitations of using 8% incidences of deformity as a reference value in all studies and those of the TSI as mentioned above call for the development of a robust deformity-based index appropriate for freshwater biomonitoring. Therefore, in the present study, we develop a new chironomid larval morphological deformity-based index called the extended toxic score index (ETSI). The ETSI included other chironomid genera, taking account of their differential sensitivity to pollution. We also demonstrated the bioindicator potential of chironomid assemblage structure in the studied river system.

2. Materials and methods

2.1. Study area description and sampling sites

The study was conducted in the Swartkops River in the Eastern Cape Province of South Africa. The river originates in the foothills of the Groot Winterhoek Mountains and flows through the towns of Uitenhage and Despatch before discharging into the India Ocean at Algoa Bay near Port Elizabeth (Fig. 1). The river catchment is mainly of marine, estuarine and fluvial origin (DWAF, 1996a). The climate is warm and temperate. The catchment receives rainfall throughout the year, with a mean annual rainfall between 300 mm–750 mm (Odume, 2011). Bushveld, fynbos and succulent thicket are the dominant natural vegetation on the Swartkops catchment (DWAF, 1996a; Kleynhans et al., 2005). The dominant invasive species include gum trees (*Eucalyptus* spp.), wattle (*Acacia* spp.), cluster pine (*Pinus* spp.) and prickly pear (*Opuntia* spp.) (Odume et al., 2012a).

Ecologically, the river supports an estuary that has the third largest inter-tidal salt mash in South Africa (Enviro-Fish Africa, 2011). The estuary is ranked eleventh most important in terms of biodiversity in South Africa (Enviro-Fish Africa, 2011). For example, it is home to about 195 bird species (Martin, 1988) and during summer, over 3500 birds can be present along the estuarine salt marsh (Enviro-Fish Africa, 2011). However, several sources of pollution, including wastewater effluent discharges, run-off from informal settlements, agricultural farms, surrounding road and rail networks, and industrial sites, severely influence the water quality of the river. Pollution from these sources is threatening the biodiversity and overall health of the Swartkops River (Odume and Muller, 2011). The Kelvin Jones wastewater treatment work (WWTW), which discharges effluent into the river at Uitenhage, has a design capacity of 24 Ml/day, treating effluent originating from both residential and industrial areas of the town of Uitenhage.

Four sites, i.e. Sites 1–4, were selected along the river. Site 1 (33° 45′ 08.4″ S, 25° 20′ 32.6″ E), in the upstream section of the river, was selected as the control site because it was less influenced by human activities. The site was situated in an area with no evident of run-offs from informal settlements, industries and agricultural activities. Site 2 (33° 47′ 29.0″ S, 25° 24′ 26.4″ E) was situated in the industrial town of Uitenhage where surrounding impacts include run-off from informal settlement, road and rail networks. The site was upstream of the effluent-discharge point of WWTW in Uitenhage. Site 3 (33° 47′ 11.8″ S, 25° 25′ 53.97″ E), also situated in the industrial town of Uitenhage, was few metre downstream of the effluent-discharge point of the WWTW. In addition to the influence of the effluent, run-off from the surrounding road and rail networks also impacted the water quality. Site 4 (33° 47′ 34.0″ S, 25° 27′ 58.7″ E), in the residential town of Despatch, was about 2.5 km downstream of Site 3 with the impact of the effluent still indicated by water chemistry variables (Odume and Muller, 2011; Odume et al., 2012b). An additional downstream site was not selected to avoid the effects of the estuary which was few kilometres downstream of Site 4. The biophysical characteristics of the four sampling sites are summarised in Table 1.

2.2. Chironomid sampling and physico-chemical variable analysis

Chironomid larvae were sampled seasonally over a period of three years between August 2009 and September 2012 according