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Macroinvertebrate community structure and feeding interactions along a pollution gradient in Gilgel Gibe watershed, Ethiopia: Implications for biomonitoring

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ABSTRACT

Feeding interactions among functional feeding groups (FFGs) of macroinvertebrates are robust indicators of aquatic ecosystem interactions. They provide information regarding organic matter processing, habitat condition and trophic dynamics. In tropical rivers with pronounced wet and dry seasons, macroinvertebrate based ecological monitoring tools are explicitly focused on metrics and indices, while ignoring interactions of FFGs. Therefore, the objective of this study was to investigate the functional feeding type metrics, diversity indices and feeding interactions among FFGs of macroinvertebrates along the water pollution gradient in Gilgel Gibe watershed, Ethiopia. Water quality parameters and macroinvertebrate community attributes were assessed for samples collected from upstream sites (15 sites), urban-impacted stretches (12 sites) and wetland-affected river zones (7 sites) of the watershed during the rainy (July) and dry (February) seasons. To understand the effect of pollution on the feeding interactions, stable carbon and nitrogen isotopes were analyzed. Macroinvertebrate-based diversity indices and functional feeding type metric showed deterioration of ecological integrity at the urban-impacted sites and substantial recovery in the wetland-affected downstream sites. Omnivorous feeding behavior of macroinvertebrates was noted for the upstream sites, whereas clear trophic guilds of FFGs were suggested for the wetland-affected river zones by the stable isotope results. The results of pollution gradient analysis and feeding interactions among FFGs revealed that the urban-impacted sites showed weaker interactions when compared to upstream and wetland influenced sites. This affirms the potential importance of feeding interactions among FFGs of macroinvertebrates in water quality monitoring.

1. Introduction

The settings of river systems are usually ideal for human settlement, which makes them prone to high human pressure. In the absence of water resources protection, rivers collect pollutants as they flow though urban areas and agricultural landscape (Devi et al., 2008; Xu et al., 2014). Recently, river pollution has become an escalating environmental problem in developing countries (Awoke et al., 2016; Ayenew, 2007; Nyenje et al., 2010). It is mainly due to unlimited population growth, which led to urbanization, intensification of agricultural activities and deforestation of river watershed, which in turn trigger the direct release of pollutants (Awoke et al., 2016; Devi et al., 2008). Several case studies reported that rivers in the developing world are being stressed and losing their natural state (Kloos and Legesse, 2010; Ndiritu et al., 2006) primarily due to anthropogenic disturbances (Tejerina-Garro et al., 2005). Therefore, monitoring and detection of the impact of environmental stressors on aquatic systems are crucial for the protection and restoration of river ecosystems through appropriate planning and implementation of integrated watershed management (Friberg et al., 2011; Iliopoulou-Georgudaki et al., 2003).

The increased production of sewage associated with the expansion of urban centers and the widespread utilization of fertilizers in agriculture within watersheds leads to the overloading of

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receiving water bodies with allochthonous nutrients, mainly nitrate and phosphate (Bannon and Roman, 2008). Such eutrophication of water bodies has been directly linked to the loss of aquatic biodiversity, functional feeding groups, and alteration of benthic food web structures and functions (Vander Zanden et al., 2005; Xu et al., 2014). Thus, aquatic ecosystems biomonitoring is very important to establish the causal relationships between stressors and ecologically substantial responses (Clements et al., 2002).

The status of aquatic life is considered as the most direct and effective measure of a water body's overall ecosystem health (i.e., chemical, physical, and biological integrity) (Nadushan and Ramezani, 2011). However, water monitoring schemes that consider the presence/absence and abundance of an indicator organism may overlook important components of the food web. Therefore, besides the composition and diversity of aquatic biota along the pollution gradient, quantitative studies of feeding relationships among FFGs of macroinvertebrates are essential to detect allochthonous material and nutrient load (Vander Zanden et al., 2005; Zah et al., 2001). In these regard, stable isotopes provide a way to trace and identify details of anthropogenic nitrogen input and pathway in water bodies (Bannon and Roman, 2008; Vander Zanden et al., 2005). The analysis of stable isotopes, which does not depend on species relative abundances, can also help us overcome sampling limitations associated with sampling efforts in river health monitoring programs (di Lascio et al., 2013). Furthermore, stable carbon and nitrogen isotopes have proven their utility to serve as a useful tool for the identification of trophic relationships among consumers and their food sources (Roussel et al., 2014; Vander Zanden et al., 2005).

Due to stepwise trophic level enrichment of isotopic signatures, stable nitrogen isotope ($\delta^{15}N$) is used to infer the trophic position, while stable carbon isotope ($\delta^{13}C$) provides information regarding the sources of energy to higher trophic level consumers (Vander Zanden et al., 2005). The nitrogen isotopic signature ($\delta^{15}N$) of aquatic primary producers, which indeed cascades to higher trophic level consumers through fractionation, is a function of concentration of dissolved inorganic nitrogen in the water (Hoefs, 2008; Loomer et al., 2015) and types of nitrogen sources (Bannon and Roman, 2008). For instance, higher inputs of inorganic nitrogen, including industrially produced ammonia, into eutrophic systems leads to more fractionation against ^{15}N, which in turn lowers the trophic position ($\delta^{15}N$) of consumers. On the other hand, low fractionation against ^{15}N in the nitrate-depleted systems leads to high nitrogen isotopic signature of aquatic consumers (Somes et al., 2010).

Macroinvertebrates are among the most common bioindicators of water pollution in tropical river systems (Aschalew and Moog, 2015; Beyene et al., 2009a, 2009b). They also occupy a central position in aquatic food web through linking food sources with top trophic level consumers (Resh, 2008). The distributions of FFGs of macroinvertebrates provide scientific information regarding organic matter processing, habitat condition, trophic dynamics (Xu et al., 2014) and ecological quality of rivers and river-associated wetlands (Ambelu et al., 2010; Mereta et al., 2013). Although a number of studies have addressed the feeding relationships among the FFGs of macroinvertebrates in the temperate region (Bunn et al., 2003; Imberger et al., 2014; Zah et al., 2001), little is known about how feeding interactions are altered by differing river water quality influenced by human activities in tropical Africa. Therefore, investigating feeding interactions among macroinvertebrates, their distribution and diversity can improve our understanding of nutrient loading and ecological integrity in aquatic ecosystems. This paper aims at assessing the macroinvertebrate community structure and feeding interactions among FFGs along a pollution gradient.

2. Materials and methods

2.1. Sampling sites

The study was conducted in the Gilgel Gibe catchment of the Omo-Gibe river basin, southwest of Ethiopia (Fig. 1). Jimma town is the largest urban and administrative center in the catchment. The small rivers, which cross Jimma town and flowing to Gilgel Gibe dam, serve as a natural sewerage lines for domestic waste (Devi et al., 2008). In the present study, a total of 34 sampling sites that permit upstream and downstream comparisons were selected from three different parts of the catchment (Reynoldson et al., 1997): upstream (Up1–Up15), urban-impacted (U1–U12) and Boye wetland/dam-affected (W1–W7) sites. The study was undertaken during the representative months of rainy (July, 2013) and dry (February, 2014) seasons.

Upstream sites (Up1–Up15) - The Rivers are relatively fast-flowing with no/little urban influence. They are partly covered with riparian vegetation.

Urban-impacted sites (U1–U12) - The sampling sites are subjected to direct urban influence from Jimma town. Domestic liquid and solid waste is directly discharged into the river from different sources. There is no/little riparian vegetation.

Boye wetland/dam-affected sites (W1–W7) - The sampling sites are downstream from Jimma town past the Boye wetland/dam. The rivers are slow-flowing.

2.2. Water sampling, processing and analysis

American Public Health Association et al. (2005) recommends in-situ measurements for the parameters that changes over time due to chemical reactions or biological changes. The levels of dissolved oxygen (DO), water temperature and electrical conductivity (EC) were measured using HACH-hd401 multi-parameter (HACH, Loveland, USA). Turbidity and current velocity (Velo) were measured using Wag-WT3020 turbidity meter (Halma, Amersham, UK) and a flow meter, respectively.

In order to collect sufficiently mixed and representative water samples across the width of rivers, inclusion of three to five points is recommended (Bartram and Ballance, 1996). Accordingly, water samples were collected from three sampling points across the width of the rivers and mixed proportionally. Unfiltered water samples were collected using 250 ml polyethylene bottles for total nitrogen (TN), total phosphorous (TP), total suspended solids (TSS) and total dissolved solids (TDS) analysis. Water samples were also filtered onsite using a filtration apparatus and Whatman glass microfiber filters (GF/F) and kept in 100 ml polyethylene bottles for the analysis of dissolved nutrients (i.e. soluble reactive phosphorus (SRP) and nitrate-nitrogen (NO_3-N)). Within six hours after sampling, samples were transported in an ice-box to the laboratory of Environmental Health Sciences and Technology, Jimma University, Ethiopia and immediately kept in a deep freezer until the analyses were made. All the aforementioned parameters were determined following the standard methods described in American Public Health Association et al. (2005). SRP and TP (after digestion with persulfate) were measured by the Ascorbic acid method. NO_3-N and TN were determined by the sodium salicylate and Kjeldahl methods, respectively. TDS and TSS were measured gravimetrically.

2.3. Macroinvertebrate sampling and identification

A rectangular frame kick net with a 250 μm mesh size on a 50 × 33 cm frame was used to collect macroinvertebrates. The river bed was disturbed with the feet of a person, facing the water current, for three minutes to dislodge macroinvertebrates found within a ten-meter stretch. After three minutes of sampling, the