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Case Study

Comparative study of diatoms and macroinvertebrates as indicators of severe water pollution: Case study of the Kebena and Akaki rivers in Addis Ababa, Ethiopia

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ABSTRACT

We assessed the relative performance of diatoms and macroinvertebrates to measure municipal and industrial impacts on the ecological integrity of the three major rivers flowing through Addis Ababa. Both community metric and multivariate statistical techniques were used to analyze the environmental variables and species data along the pollution gradient. This study in the Addis Ababa urban area revealed that three biologically highly stressed rivers are being impacted primarily by physical habitat degradation and both point and nonpoint pollution. The macroinvertebrate composition was liable to severe physical habitat and chemical water quality degradation. Consequently, macroinvertebrates were less diverse and not found at all at the most polluted sites with very low dissolved oxygen levels. Based on community metrics and multivariate analysis results, diatoms more reliably indicated a gradient of pollution than macroinvertebrates. However, both organism groups equally discriminated the two relatively unimpacted upstream sites from all other impacted sites. As diatoms are immobile and ubiquitous (i.e., at least a few can be found under almost any condition), they are good indicators of pollution levels among heavily impacted sites where macroinvertebrates are completely absent or less diverse. Therefore, diatoms are the powerful bioindicators for monitoring urban-impacted and seriously stressed rivers and to examine pollution gradients and impacts of specific pollution sources.

1. Introduction

Aquatic ecosystems are threatened world-wide by pollution, as well non-sustainable land-use and water-management practices that are reaching critical levels (Mayes et al., 2007; Devi et al., 2008). Studies of the integrity of water bodies have evolved from studies of water quality monitoring based primarily on water chemistry (Cairns, 1995) to more comprehensive assessments of aquatic ecological systems (Hering et al., 2006; Astin, 2007). Studies to generate information on

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water quality degradation and particularly on its impact on biodiversity have been carried out in many countries, including countries in the tropics (Ndiritu et al., 2003; Harding et al., 2005). Biological parameters are increasingly studied as more sensitive indicators of ecosystem integrity than physicochemical parameters (Craft et al., 2007; Flinders et al., 2008; Smith et al., 2007).

The usefulness of diatoms in evaluating present and past conditions of water quality and environmental change is increasingly being recognized world-wide, resulting in renewed interest in using these organisms in surface water quality monitoring (Ndiritu et al., 2003; Potapova and Charles, 2007; Taylor et al., 2007; Tison et al., 2007). The long history of diatom research has resulted in the development of basic conceptual and analytical approaches to monitor environmental conditions using autecological and ecological indices (Sabater, 2000; Harding et al., 2005; Lane and Brown, 2007; Potapova and Charles, 2007). Several authors consider peak diatom biomass and species composition to be valuable parameters of environmental disturbances in freshwater ecosystems. Studies of diatoms in the Nairobi River in Kenya and in South Africa rivers demonstrated that diatoms can be effectively used in the development of suitable watershed management tools (Ndiritu et al., 2003; Harding et al., 2005).

Studies of aquatic invertebrates are influenced by flood and drought, hydrology, substrate, habitat, and discrepancies in food availability and seasonal variations in invertebrate occurrence (Miller and Golladay, 1996). Invertebrates are also weak indicators of euthrophication and diffuse and point source impacts which may be identified by direct measurements of diatom associations (Passy et al., 2004; Harding et al., 2005). Unlike macroinvertebrates and fish, diatom species occur in a wider variety of waters (at least a few can be found under almost any condition) and their distribution is cosmopolitan (Potapova and Charles, 2007). Diatoms are also suited for monitoring very heavily impacted systems where other types of organisms are absent (Taylor et al., 2007). The study of diatoms for monitoring the integrity of rivers has increased because of the limitations of macroinvertebrates and fish as indicators and significant improvements in technologies for diatom assessment that increase the information per cost ratio (Charles et al., 2002; Potapova and Charles, 2007).

Few comparative studies using diatoms and macroinvertebrates were carried out in temperate regions, thereby revealing that both organism groups were not much correlated in terms of diversity or ecological quality indicator values. Such findings were reported from the Woluwe river in Western Europe (Triest et al., 2001), from Finnish rivers and streams in northern Europe (Soininen and Könönen, 2004) and at European level (Hering et al., 2006). On the other hand, different authors reported that both groups of organisms provide complementary information on environmental qualities (Johnson et al., 2006; Feio et al., 2007). However, no comparative studies have been carried out of diatoms and macroinvertebrates in disturbed and stressed rivers systems towards developing tools that can be used by water quality managers in developing countries from tropical regions. This study investigated the responses of diatom and macroinvertebrate community structures to major environmental gradients in three disturbed and stressed rivers in Addis Ababa, Ethiopia, using indirect ordination and comparing the estimated ecological state and intra-site variation found by two assessment methods.

2. Material and methods

2.1. Study area and location of sampling sites

The study area is located in Addis Ababa, the capital of Ethiopia, which had a projected population of 3.06 million in 2007 (CSA, 1998). Located in the central highlands between 2200 and 2600 m altitude, the climate of this city is temperate and subhumid. The Kebena, Big Akaki and Little Akaki rivers originate on nearby Entoto Mountain and flow through the city from north to south (Fig. 1). All of them serve as natural sewerage lines for domestic and industrial wastes. According to the Ethiopian Central Statistical Agency (CSA, 2006), about 51.9% of all Ethiopian large and medium scale industries are located in Addis Ababa. It was indicated that 80 and 20% of the raw or inadequately treated industrial effluents are discharged into the Little Akaki and Akaki rivers, respectively (Abate, 1994). Several studies identified these two rivers as the most polluted water courses in Ethiopia (Alemayehu, 2001; ESTC, 2004).

Sixteen sites were sampled along the courses of the Kebena, Little Akaki and Akaki rivers. The sampling sites labeled as K (for the Kebena), L (for the Little Akaki) and B (for the Big Akaki) rivers are numbered in ascending order downstream along the pollution gradient (Fig. 1). All sites were sampled once between 20 and 25 May 2007, before the big rainy season.

2.2. Habitat survey

The condition of local stream and river habitats, also known as the habitat template, influences the structure and organization of biological communities (Downes et al., 1995). Information on habitat parameters facilitates the assessment of the spatial and temporal distribution of organisms and underlying factors (Maddock, 1999). The qualitative habitat evaluation index (QHEI) is a physical habitat index designed to provide an empirical, quantitative evaluation of the general lotic macrohabitat characteristics that are important to aquatic communities (Rankin, 2006). The QHEI is composed of six principal metrics, i.e., substrate, instream cover, channel morphology, riparian zone, pool quality, riffle quality and gradient (Rankin, 2006), with a maximum possible QHEI site score of 100. Each of the metrics was scored individually and then summed to provide the total QHEI site score.

2.3. Water sampling and analysis

Sampling at 3–5 points is usually sufficient and fewer points are needed for narrow and shallow rivers and streams (Bartram and Balance, 2001). Thus a composite sampling technique was employed to take water samples at 3 points across the width of the rivers for chemical analysis. The water samples were collected by inserting clean bottles of 2 l to a 30-cm depth in the opposite direction of the current flow of the

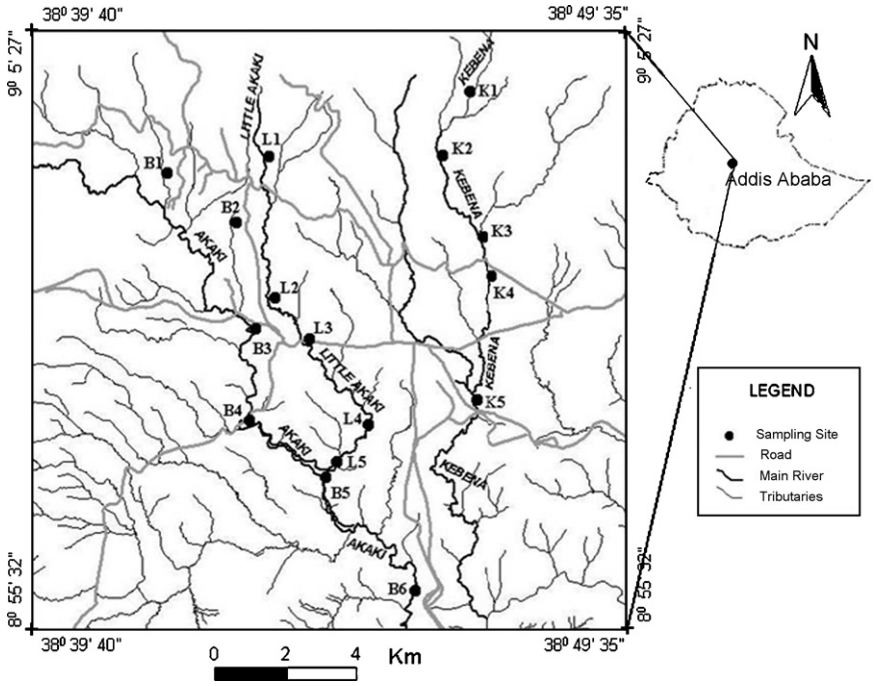

Fig. 1 – Study area, drainage, location of sampling sites on the Kebena (K), Little Akaki (L) and Akaki (B) rivers. Sites are consecutively numbered in the direction of river flow.

rivers. The samples were transported to the laboratory of the Limnology Department, Addis Ababa University, within 1–6 h for analysis and chemical analysis following the standard methods (APHA, 1998), i.e., indophenol blue, molbidosilicate, stannous chloride, phenoldisulphonic, titrimetric and oxygen probe methods were used for NH_3–N + NH_4–N, SiO_2, soluble reactive phosphorous (SRP), NO_3–N, alkalinity and BOD_5, respectively. Total dissolved solids (TDS) and pH, conductivity, salinity, DO, and temperature were measured in situ using HACH Pocket Pal^{TM} TDS Tester and CX-401 multifunctional pH, conductivity, salinity, DO, pressure, and temperature data logger, respectively.

2.4. *Diatom sampling, identification and counting*

Three natural substrates (stones) were sampled randomly at each sampling site within 10 m reach and a total surface area of 25 cm^2 substrates scraped with a knife and pooled to form a single sample, as recommended by Kelly et al. (1998). The samples of diatoms were preserved with 4% (m/m) formalin and sent to the Laboratory of Plant Science and Nature Management (APNA) of the Vrije Universiteit Brussels, Belgium, for processing and taxonomic analysis. Diatom frustules were cleaned with concentrated sulfuric acid and nitric acid. Then acid-cleaned samples were filled to a volume that provides adequate density of diatom valves and mounted on slides with styrax mounting medium. Diatom frustules were counted with a Leitz Dialux 20 EB light microscope at 1000× magnification and only valves with more than half the intact valves within the field were counted. A total of 500 diatom valves were identified and counted in random fields using standard taxonomic keys (Gasse, 1986; Krammer and Lange-Bertalot, 1986, 1988, 1991a,b; Kelly, 2000; John et al., 2003; Round et al., 2007). Finally, the number of cells per mm^2 was calculated using a formula adapted and modified from USGS Protocol P-13-52 (Charles et al., 2002) and then the relative abundance within sites was calculated.

$$\frac{\text{Cells}}{\text{mm}^2} = \frac{C * Atf * Df * Vt}{FA * Nfs * Vsc * Ats}$$

where C: total diatom cells counted, Atf: area of the total fields on the microscope slide cover, Df: dilution factor of the sample, Vt: volume of total sample, FA: microscopic field area, Nfs: number of microscopic random fields scanned, Vsc: volume of subsample used for counting, Ats: total area of substrate scrapped for diatom sampling.

2.5. *Macroinvertebrate sampling, identification and counting*

Collections of macroinvertebrates from more than one habitat type may introduce variation that can potentially mask water quality differences among sites (Jeffrey and Charles, 2003). To minimize this variation, all samples were collected from the same habitat types of riffle zones of streams in areas where there was the best canopy coverage and side bank macrovegetation. Riffle communities of streams are also more diverse in invertebrate forms than pools (Gerth and Herlihy, 2006). A rectangular frame kick net with a 500 μm net on a 50 cm × 33 cm frame was used for collection. The kick net was placed in the river of different depths in the opposite direction of current flow. The riverbed was agitated continuously with the feet of the same person for 3 min to dislodge macroinvertebrates within a 10-m stretch and allow the current to

sweep them into the net. After 3 min of sampling the contents of the net were transferred into a collecting jar containing 90% ethanol. This procedure was repeated 3 times per site to calculate mean values. Jars were transported to the laboratory of the Limnology Department, Addis Ababa University, for sorting and dissecting. In the laboratory, the contents were transferred into a bowl, sufficient water was added and the supernatant was poured through a sieve (500 μm pore size) to retain the macroinvertebrates. This was repeated until all macroinvertebrates were separated from the sand and mud. All macroinvertebrates were sorted and then identified using a binocular dissecting microscope and assigned to their respective taxonomic levels. For monitoring activities most studies used species identification for diatoms (Sabater, 2000; Ndiritu et al., 2003; Potapova and Charles, 2007; Taylor et al., 2007) and family or higher level identification for macroinvertebrates (Hailu and Legesse, 1997; Griffith et al., 2001; Hering et al., 2006; Flinders et al., 2008). Genus level identification of macroinvertebrates does not offer substantial advantage over family level identification and species level identification appeared to be unnecessary for monitoring programs (Bowman and Bailey, 1997; Hewlett, 2000; Waite et al., 2004).

2.6. Data analysis

Relationships between the environmental and biological (macroinvertebrates and diatoms) data were assessed using canonical multivariate analysis with the software program CANOCO 4.5 (ter Braak and Šmilauer, 2002). Detrended correspondence analysis (DCA) was used to determine the appropriate response model (linear or unimodal) for both the invertebrate and diatom data. The performed DCA gives a gradient length < 2 standard deviations (S.D.s) in both cases, implying that taxa abundance exhibit linear response to environmental gradients (ter Braak and Šmilauer, 2002). Therefore, we first used principal component analysis (PCA) to reduce complexity in the sample data on the basis of community composition alone. Then we used redundancy analysis (RDA) to explicitly investigate the relationship between community composition and environmental variables. During the RDA analysis the species scores were post transformed and divided by the standard deviation to standardize the ordination diagram for species data and correlation instead of covariance (ter Braak and Šmilauer, 2002). Species abundance data were log transformed [(log $(x + 1)$] before the RDA analysis to prevent extreme values (outliers) from unduly influencing the ordination (ter Braak and Šmilauer, 2002).

RDA analysis involved a forward selection procedure to obtain the smallest set of variables explaining statistically significant variance in the community data a procedure that selects at each step the predictor variables that contribute most to the explained variance in the response variables. Significance was tested using the Monte Carlo test with 999 permutations (ter Braak and Šmilauer, 2002). In the preliminary analysis environmental variables with an inflation factor larger than 5 were removed. An inflation factor of 5 and higher has been identified as an indicator of colinearity in multivariate analysis (Griffith et al., 2001).

The Monte Carlo permutation under the reduced model method better maintains the type I error in small data sets (ter Braak and Šmilauer, 2002). This model was used to test the statistical significance of the relation between species distribution and the whole set of environmental variables based on the first and sum of all canonical eigenvalues. The resulting test determines the significance of the first canonical ordination axis and that of all canonical axes together.

The software packages OMNIDIA, Version 3 and BioDiversity Professional, Version 2 were used to calculate diatom and macroinvertebrate indices, respectively. Indices (Diversity, Simpson, Alpha, %PT, %DT and %EPT) are currently getting acceptance and are widely used (Hering et al., 2006; Johnson et al., 2006). They can be applied to a range of factors (physical, chemical, and biological) that stress biological systems, and they are relatively easy to measure and interpret (Karr and Chu, 1999). Our data showed that only few families of macroinvertebrate taxa at almost all sites, which limits us not to use the macroinvertebrate biotic indices in this study. STATISTICA, Version 7.0 was also used to test the correlation among the environmental variables and biological metrics.

3. Results

3.1. Physicochemical and habitat qualities

Physicochemical parameters and QHEI values indicated a sharp loss of ecological quality of rivers downstream with increasing pollution load. The most significant parameter related to sustainability of aquatic life in rivers, Dissolved oxygen (DO) was depleted to a level of 0.1 mg/l in downstream sites (Table 1). Biochemical oxygen demand (a measure of organic pollution and a cause for oxygen depletion) reached to a level of 1250 mg/l, i.e., about 400-folds above the normal level (3 mg/l). Other nutrients (phosphate, nitrate and ammonia plus ammonium were also elevated in downstream sites along the pollution load (Table 1). The habitat quality, which was measured by the QHEI and classes based on Rankin (2006) revealed that most sites were within poor quality classes and indicative of high habitat degradation and loss in the local rivers (Table 1).

3.2. Diatom metrics

In the accepted RDA model (Fig. 2), redundancy analysis of the metric data with stepwise selection reduced the environmental data to 8 variables (DO, salinity, silicon dioxide, SRP, TDS, nitrate, pH and QHEI). The statistical significance of the relationship between the diatoms and the whole set of environmental variables based on the first and the sum of all canonical eigenvalues was found to be significant ($p < 0.05$) for the first axis as well as for all canonical axes together (Fig. 2). The RDA-triplot of samples, diatoms and environmental variables based on the first two axes explained 41.8% of the variance in the diatom species data, 61.5% of the variance in the fitted species data, and 61.5% of the variance in the correlated and class means of species with respect to the environmental variables. The eigenvalue of axis 1, 2, 3 and 4 were 0.239, 0.179, 0.081 and 0.065, respectively. The scale in S.D. units was −1 to 1 for both diatoms and environmental variables (see full names of diatom codes in Appendix A). In

Table 1 – Spatial variation of environmental variables along the Kebena (K), Little Akaki (L) and Akaki (B) rivers

Variables	K1	K2	K3	K4	K5	L1	L2	L3	L4	L5	B1	B2	B3	B4	B5	B6
Temperature (°C)	14.0	22.0	21.7	20.5	15.0	15.5	17.1	18.6	18.9	19.3	23.3	17.3	18.9	20.9	18.4	15.3
Conductivity (μS/cm)	105	674	613	538	659	526	608	741	672	972	430	840	1130	870	628	1200
Alkalinity (mg/l as $CaCO_3$)	95	135	170	213	305	115	165	235	280	295	156	210	295	345	291	316
NH_3–N + NH_4–N (mg/l)	0.14	3.3	7.2	5.4	10.9	1.8	4.6	10.4	14.6	7.8	1.3	7.4	6.1	5.9	12.7	7.9
pH	6.84	5.78	5.80	5.76	7.60	7.45	7.12	7.36	7.45	7.02	6.95	5.90	5.85	6.00	6.92	7.56
DO (mg/l)	7.8	5.8	1.9	3.0	1.7	6.5	1.8	0.7	0.1	3.3	7.3	1.0	4.3	6.8	3.8	2.1
TDS (mg/l)	315	392	385	382	395	333	492	429	448	561	279	593	709	541	324	810
Salinity (PSS)	0.06	0.34	0.36	0.4	0.34	0.27	0.31	0.38	0.34	0.49	0.21	0.42	0.54	0.45	0.32	0.62
SiO_2 (mg/l)	12.8	12.4	19.0	19.9	61.7	17.1	50.3	25.0	29.8	53.8	22.3	63.7	21.3	15.7	16.7	29.8
NO_3–N (mg/l)	0.21	1.72	4.85	5.32	3.21	0.48	6.74	9.8	5.28	12.5	0.34	1.78	8.23	5.65	16.2	18.2
SRP (mg/l)	0.00	0.21	0.47	0.41	1.0	0.22	0.96	0.97	1.1	1.2	0.17	3.1	3.8	2.5	3.8	4.5
BOD_5 (mg/l)	2.8	153	657	153	712	42	591	832	1250	684	23	1176	413	98	769	1138
QHEI	30.6	43.3	48.7	42.7	48.7	42.2	48.0	59.3	44.3	55.3	52.7	44.1	62.7	54.7	54.0	59.0
Altitude (m)	2622	2501	2420	2389	2167	2556	2364	2327	2271	2209	2532	2495	2450	2253	2222	2154

Note: Practical salinity scale (PSS) synonymous with practical salinity unit (PSU) was used to measure salinity as a conductivity ratio with no units. BOD_5: 5 days biochemical oxygen demand; DO: dissolved oxygen; QHEI: qualitative habitat evaluation index; SRP: soluble reactive phosphorus; TDS: total dissolved solids.

this ordination, the metric environment correlation for the four axes 1–4 was strong (i.e., 0.956, 0.980, 0.926 and 0.890, respectively). The first axis revealed a gradient primarily associated with habitat quality. The environmental variable best correlated to this axis, the qualitative habitat evaluation index (QHEI), indicated a gradient of suboptimal to poor macrohabitat quality (Table 1). The second canonical axis described the pollution gradient of high dissolved oxygen in relation to high nutrient enrichment and pH increase. This axis could be further explained by scrutinizing two variables (BOD_5 and ammonium plus ammonia), which were excluded due to colinearity.

Among the abundant diatom species commonly found at the sampling sites with a sensitivity value of 1 was *Nitzschia palea* (NPAL), strongly correlated with the second negative RDA axis. Species assemblages were associated with high nutrient enrichment and were most abundant at the polluted sites B5, B6, L4 and L5. *Achnantidium minutissima* (AMIN), a common taxon and low nutrient indicator (Potapova and Charles, 2007), was strongly correlated with the first RDA positive axis and was most abundant at the upstream site K1. *Navicula frugalis* (NFRU), one of the most common species found at all sites, was strongly correlated with the first RDA negative axis and was the dominant species at sampling sites B2, B3 and K3. In general, RDA ordination of the diatom species and environmental variables classified the sampling sites (Fig. 2) in ascending order of pollution level from the relatively clean upstream sites to the progressively more polluted downstream sites (K1 → B1 → K2 → K3, K4, B2, B3, B4, L1, L2 → K5, L3 → B5, B6, L4, L5).

3.3. *Macroinvertebrate metrics*

In the accepted RDA model (Fig. 3), redundancy analysis of the metric data with stepwise selection reduced the environmental data to 6 variables (BOD_5, salinity, TDS, nitrate, pH and QHEI). The statistical significance of the relationship between macroinvertebrate families and the selected set of environmental variables based on the first and sum of all canonical eigenvalues was found to be significant ($p < 0.05$) for both the first axis and all canonical axes together (Fig. 3). The RDA-triplot of samples, macroinvertebrates and environmental variables based on the first two axes explained 47.8% of the variance in the invertebrate data, 73.1% of the variance in the fitted invertebrate data, and 73.1% of the variance in the correlated and class means of invertebrates with respect to the environmental variables. The eigenvalue of axis 1, 2, 3 and 4 were 0.276, 0.203, 0.095 and 0.069, respectively. The scale in S.D. units was −1 to 1 for both macroinvertebrates and environmental variables (see full names of invertebrate codes in Appendix B). In this ordination, the metric environment correlation for the four axes (1, 2, 3, and 4) was strong (i.e., 0.912, 0.9750, 0.755 and 0.822, respectively).

The first axis of the RDA ordination (Fig. 3) revealed a gradient primarily associated with pollution and macrohabitat quality. The environmental variable best correlated with this axis (nitrate) followed a high nutrient enrichment gradient and was also strongly correlated with the pollution-tolerant Oligochaetes taxon. This axis could be further explained by scrutinizing the other variables (SRP, and ammonium plus

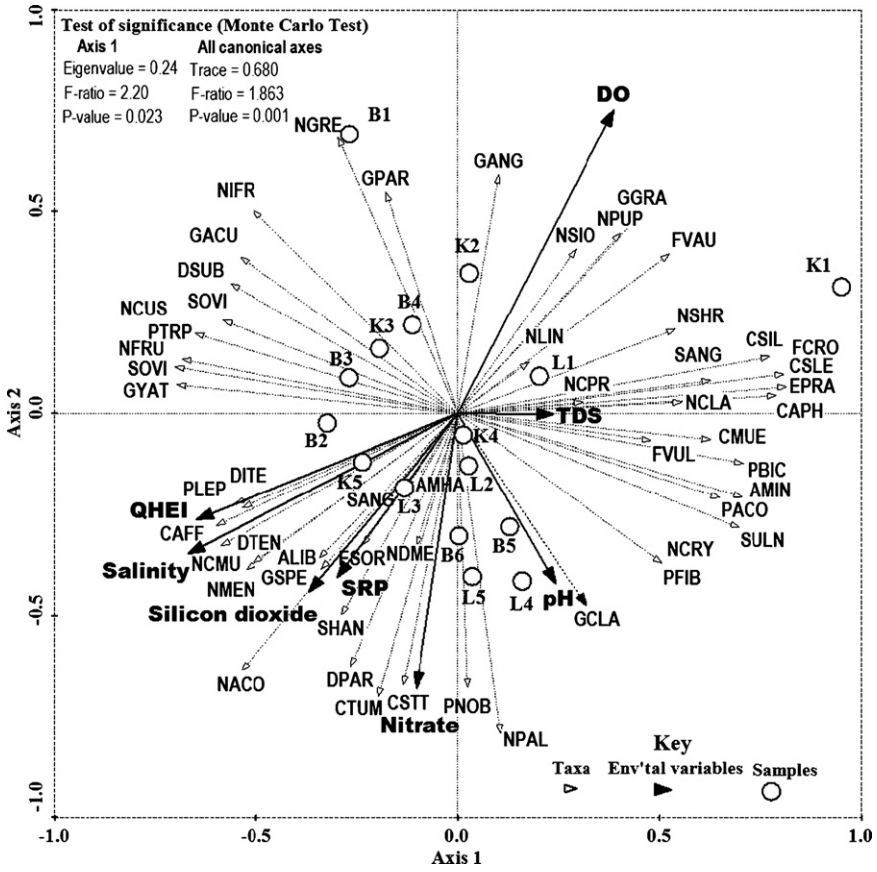

Fig. 2 – RDA-triplot of samples, diatoms and environmental variables based on the first two axes (see full names of species codes in Appendix A).

ammonia), which were excluded because of colinearity. The second canonical axis described the salinity gradient. Although not clear and consistent with the environmental data, RDA ordination of the macroinvertebrate families and environmental variables classified the sampling sites in ascending order of pollution levels from the upstream sites (relatively better quality) to the highly polluted downstream sites (K1, B1 → B3, B4, K2, K3, K4, L1, L2, L3, K5 → B2 → B5, B6, L5). L4 was excluded from the RDA ordination diagram (Fig. 3) because of the complete absence and impairment of invertebrate fauna and hence it was categorized as the most polluted site.

3.4. *Diatom metrics and indices*

Of the more than 500 individuals counted at all sampling sites, a maximum of 30 species were found at the upstream site K1 and a minimum of 4 different species at the downstream site B6. Similarly, the species diversity index, which measures both richness and evenness of distributions, revealed that species diversity was higher at the upstream sites B1, K1 and L1 (Table 2) than their corresponding downstream sites. A clear gradient towards gradually lowered diversity was observed in the Kebena river and the Little Akaki river, but was not gradually for the Akaki river due to B4 as it is diluted by clean water from its tributaries coming from unimpacted sites. This pattern is further reflected in the Simpson and Alpha diversity indices (Table 2). The specific pollution-sensitive index (IPS), which is widely used as a preferred index to monitor river water quality (Hering et al., 2006), also indicated relatively better ecological quality in the upstream sites than the corresponding downstream sites, similar to the diatom biological index (IBD). However, K1 appeared to be the least affected site (IPS $>$ 11) whereas other upstream sites (B1, L1) reached IPS values lower than 6 (Table 2).

3.5. *Macroinvertebrate metrics and indices*

A maximum of 5 and a minimum of 1 families of macroinvertebrate were found at individual study sites. Macroinvertebrates were completely absent at sampling site L4 on the Little Akaki River (Table 3), where the abbattoir waste discharge their untreated effluents but gradually recovered at sampling site L5 downstream as it was diluted by its tributaries and natural self-purification. Unlike the diatoms, macroinvertebrate diversity indices (both Simpson and Alpha) failed to distinguish the upstream sites from the corresponding downstream sites on the basis of diversity (Tables 2 and 3). The percentage of pollution-tolerant taxa (%PT) was found to be greater than 85% at all sampling sites (Table 3). However, the upstream sites K1 and B1 exhibit the highest %EPT, thereby strongly discriminating from all other sites. Simpson, Alpha and %PT indices of macroinvertebrates were significantly

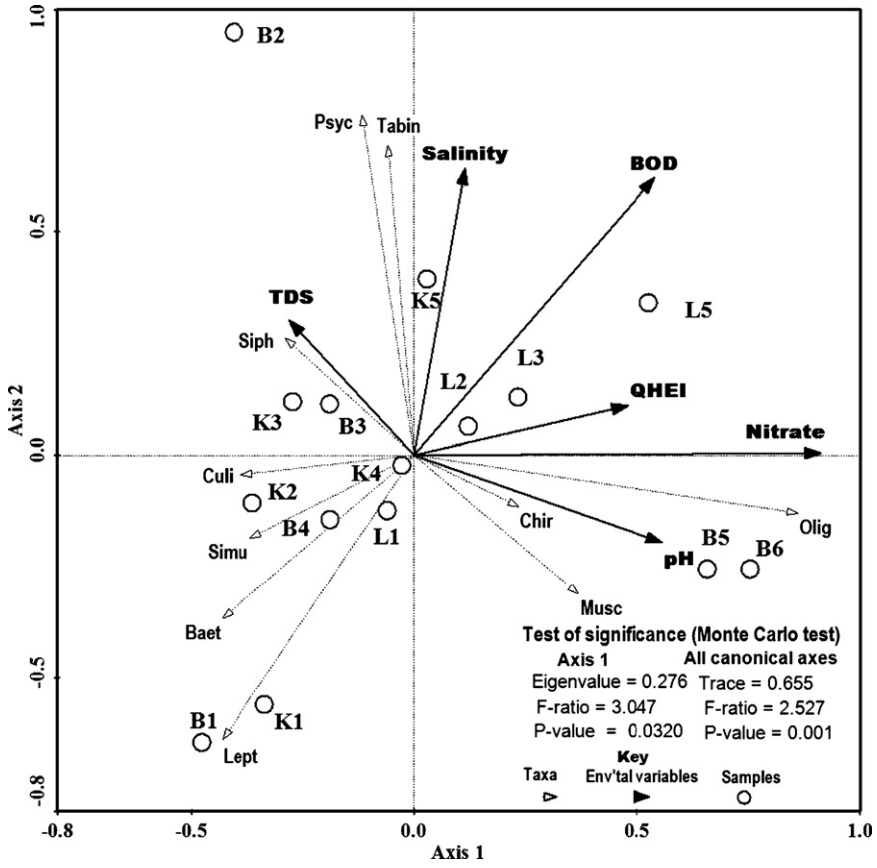

Fig. 3 – RDA-triplot of samples, macroinvertebrates and environmental variables based on the first two axes (see full names of family codes in Appendix B).

correlated (Table 4). The Spearman rank correlation between macroinvertebrate and diatom diversity (Alpha and Simpson) and percent pollution-tolerant taxa (%PT) was very weak and insignificant ($r = 0.346$ and $p > 0.05$).

4. Discussion

Accelerated pollution and eutrophication of rivers and reservoirs because of human activity are a concern throughout the world (Jonnalagadda and Mhere, 2001; Burcher and Benfield, 2006); including Africa example from Kenya (Ndiritu et al., 2003) and Ethiopia (Hynes et al., 1989; Hailu and Legesse, 1997; Alemayehu, 2001; Devi et al., 2008). Unlike in developed nations, where stringent regulations have been implemented to restrict the discharge of untreated wastewater into rivers and streams, existing pollution legislation in developing countries, including Ethiopia, is weak and generally not adequately enforced (Kumie and Kloos, 2006). Ethiopian rivers and streams flowing through larger communities become heavily polluted because they are widely used for domestic, commercial, and industrial purposes (Hailu and Legesse, 1997; Alemayehu, 2001). In this study the chemical variables revealed that high organic pollution load reached the maximal level ($BOD_5 = 1250$ mg/l) and depletion of dissolved oxygen from 7.8 mg/l to a level of 0.1 mg/l in downstream sites. The specific pollution-sensitive index (IPS) which is used as a preferred index to monitor river water quality in many studies (example, Hering et al., 2006) classified all the sampling sites as either moderate, poor or bad, with no sampling site in the good or high quality categories.

A study conducted about 20 years ago on the same study area by Hynes et al. (1989) confirmed that even in polluted sites downstream there were pollution-sensitive macroinvertebrate taxa. However, in this study no macroinvertebrate fauna was found in the Little Akaki River at sampling site L4, where the abbatoir waste directly discharged without adequate treatment in addition to its bedrock formation and high turbulence. But immediately downstream of L4 at the sampling site L5 macroinvertebrates retrieved back might be as result of dilution from its tributaries and natural self-purification. And only 1 family of chironomidae at the Akaki River site B6 downstream, which reflects increasing pollution of the local rivers in recent years. The untreated or inadequately treated industrial wastes, sewage, solid and liquid wastes from households and commercial firms dumped into the numerous streams in Addis Ababa end up in the Akaki and Kebena rivers, which exacerbate the local river pollution problem. Even the less polluted upper most sampling sites B1, K1 and L1 were poor in macroinvertebrate diversity and more than 85% of the macroinvertebrates found there were dominated by pollution-tolerant taxa. Miller and Golladay

Table 2 – Spatial variation of diatom metrics along the Kebena (K), Little Akaka (L) and Akaki (B) rivers

Indices	K1	K2	K3	K4	K5	L1	L2	L3	L4	L5	B1	B2	B3	B4	B5	B6
Abundance	718	624	601	654	624	592	558	584	588	616	526	500	719	543	509	523
Richness	30	11	16	10	10	17	8	6	11	9	17	8	8	14	8	4
Evenness	0.74	0.65	0.41	0.54	0.57	0.61	0.79	0.74	0.54	0.30	0.64	0.51	0.35	0.60	0.36	0.53
Diversity	3.64	2.25	1.63	1.81	1.90	2.50	2.38	1.91	1.87	0.95	2.61	1.54	1.04	2.27	1.08	1.05
Simpson	1.58	1.04	1.20	1.00	1.00	1.23	0.90	0.78	1.04	0.95	1.23	0.90	0.90	1.15	0.90	0.60
Alpha	6.33	1.9	3.02	1.68	1.69	3.27	1.33	0.93	1.92	1.50	3.36	1.35	1.26	2.63	1.35	0.59
IPS	11.8	5.2	5.2	3.9	2.7	5.2	5.4	4.0	2.2	1.3	5.9	4.2	4.7	5.8	1.7	1.5
IBD	13.6	8.6	6.9	7.2	6.6	9.0	8.3	6.8	5.3	6.0	7.7	5.4	5.9	8.5	6.8	6.1
% PT	29.9	34.8	19.5	71.1	52.2	73.0	59.1	50.9	56.6	83.3	63.1	18.6	8.3	45.5	85.9	74.4

Note: IPS: specific pollution-sensitive index, IBD: diatom biological index, % PT: percent pollution-tolerant taxa.

Table 3 – Spatial variation of macroinvertebrate metrics along the Kebena (K), Little Akaki (L) and Akaki (B) rivers

Indices	K1	K2	K3	K4	K5	L1	L2	L3	L5	B1	B2	B3	B4	B5	B6
Abundance	83	335	564	491	529	167	142	413	265	105	52	460	705	295	343
Richness	2	5	4	3	4	3	5	3	4	5	4	2	1	3	2
Evenness	0.53	0.15	0.03	0.03	0.12	0.16	0.26	0.04	0.14	0.25	0.62	0.09	0.00	0.27	0.26
Simpson	0.21	0.09	0.01	0.01	0.06	0.07	0.19	0.02	0.10	0.18	0.46	0.02	0.00	0.02	0.09
EPT/Chir	0.14	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.03	0.09	0.03	0.01	0.00	0.00	0.00
Alpha	0.37	0.84	0.58	0.43	0.59	0.52	1.01	0.44	0.67	1.09	1.01	0.27	0.12	0.47	0.28
% EPT	12.1	1.8	0.2	0.2	0.0	1.8	0.0	0.0	0.0	8.6	1.9	1.1	0.0	0.0	0.0
% DT	88.0	95.5	99.5	99.6	96.8	96.4	89.4	99.3	94.7	90.5	71.2	98.9	100	92.2	95.6
% PT	88.0	96.4	99.8	99.8	99.8	98.2	98.6	99.8	99.3	91.4	82.7	98.9	100	99.3	100
% Diptera	88.0	98.2	99.8	99.8	99.8	96.4	91.6	99.5	97.4	91.4	98.1	98.9	100	92.9	95.6

Note: % Diptera: percent (mosquitoes, midges, flies); % DT: percent dominant taxa; % EPT: percent (Ephemeroptera + Trichoptera + Plecoptera); EPT/Chir: ratio of EPT to Chironomidae; % PT: percent pollution-tolerant taxa.

Table 4 – Spearman rank-order correlations among the three selected common diatom and macroinvertebrate indices for all sampling sites excluding L4 (the marked correlations are significant at $p < 0.05$)

Indices	Diatom			Invertebrate		
	Simpson	% PT	Alpha	Simpson	% PT	Alpha
Invertebrate						
Simpson	0.015	0.052	−0.014	1.000	−0.749*	0.584*
% PT	−0.331	−0.346	0.276	−0.749*	1.000	−0.558*
Alpha	0.112	0.171	0.073	0.584*	−0.558*	1.000
Diatom						
Simpson	1.000	0.990*	−0.155	0.015	−0.331	0.112
% PT	0.990*	1.000	−0.123	0.052	−0.346	0.171
Alpha	−0.155	−0.123	1.000	−0.014	0.276	0.073

(1996) found that floods and drought caused a 90% reduction in macroinvertebrate densities. In Ethiopia, major flash floods and long dry spells might reduce the abundance and distribution of macroinvertebrates, in addition to the intensive human impact.

The presently preferred use and value of the aquatic invertebrate method is limited by hydrology, substrate(s), habitat, food availability, seasonal and spatial variations and the severe human impact in recent years. The invertebrate indices also do not provide reliable information on eutrophication, and diffuse and point source impacts, which have been identified by direct measurements of diatom associations (Sabater, 2000; Harding et al., 2005; Duong et al., 2007). This study confirmed that the diversity indices of the invertebrate taxa (Alpha and Simpson) failed to indicate the pollution gradient downstream. For diatoms, by contrast, separation of communities among sampling sites was clear but the corresponding macroinvertebrate communities were more similar to each other and were dominated by pollution-tolerant taxa. A weak correlation between macroinvertebrate and diatom diversity and their pollution indices was insignificant in these three Ethiopian rivers, a finding similar to those reported by Triest et al. (2001) of the Woluwe River in Brussels and by Soininen and Könönen (2004) of Finnish rivers and streams.

The combined effects of these environmental variables created habitat conditions that eliminated most of the pollution-sensitive river fauna. As illustrated by the RDA analysis, both metrics were similarly sensitive in indicating the pollution gradient; relatively better water quality at the upstream sites and much degraded downstream sites to a level of zero. Only species belonging to the families Chironomidae and Tubificidae were abundant at the urban-impacted sites, where the total number of taxa was only five or less but with abundance levels up to 705. This pattern is a characteristic of a severely disturbed aquatic ecosystem. Organisms physiologically adapted to low oxygen tension exploit the excess nutrients available and thus dramatically increase in abundance. Families belonging to the Ephemeroptera are clear-water fauna (Bouchard, 2004) and disappear as pollution load increases.

Once oxygen levels are too low, the macroinvertebrate communities are less relevant to indicate a gradient, whereas diatom assemblages still show a gradient in diversity and in pollution indices across the severely polluted sites. Although both diatoms and macroinvertebrates responded to eutrophication/organic and industrial pollution gradients, diatom metrics were thus relatively robust to show the gradient under heavy pollution and water quality degradation, even where invertebrates were absent. Duong et al. (2007) also reported that diatoms were found robust in responding for the urban pollution and classifying the river water quality classes in Vietnam.

5. Conclusion

This study in the Addis Ababa urban area revealed three biologically highly stressed rivers impacted primarily by physical habitat degradation and both point and nonpoint pollution. The macroinvertebrates composition was more liable to physical habitat and chemical water quality than diatoms and macroinvertebrates were not found at all at the most polluted sites with very low oxygen levels. Based on indices and multivariate analysis results, diatoms more reliably indicated a gradient of pollution than macroinvertebrates. Both organism groups, however, equally discriminated the two relatively unimpacted upstream sites from all other impacted sites. As diatoms are immobile and ubiquitous (i.e., at least a few can be found under almost any condition), they are good indicators of pollution levels among heavily impacted sites where macroinvertebrates are absent. Therefore, diatoms are the preferred bioindicators for monitoring urban-impacted and seriously stressed rivers and to examine pollution gradients and impacts of specific pollution sources. Detailed studies that consider seasonal variations and the factors in diatom composition are recommended in order to develop a refined diatom-based river water quality monitoring tool in the region that will guide and support sound management decisions. Lastly, responsible authorities need to take urgent ameliorative and preventive measures to improve the ecological integrity of these rivers as part of efforts to restore their ecology and reduce public health risks within the urban area and downstream river stretches.

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Appendix A

Diatom species found in the three rivers with their Omnidia code, and sensitivity (*s*) and indicator (*v*) values (Kelly, 2000)

Omnidia code	Species name	GDI/IPS	
		s	*v*
AMIN	*Achnanthes minutissima* Kützing v.minutissima	5	2
AMHA	*Amphora hartii* Cholnoky	2.6	2.2
ALIB	*Amphora libyca* Ehrenberg	2.6	2.2
CSIL	*Caloneis silicula* (Ehrenberg) Cleve	3.8	2.6
CSTT	*Cyclotella stelligera* (Cleve et Grunow)	2.9	1
CAFF	*Cymbella affinis* Kützing	4.7	2.6
CAPH	*Cymbella amphicephala* Naegeli	4.7	2.6
CMUE	*Cymbella muelleri* Hustedt	4.7	2.6
CSLE	*Cymbella silesiaca* Beleisch	4.7	2.6
CTUM	*Cymbella tumida* (Brebisson) Van Heurck	4.7	2.6
DSUB	*Denticula subtilis* Grunow	3.7	2.3
DTEN	*Denticula tenuis* Kützing	3.7	2.3
DITE	*Diatoma tenuis* Agardh	3.9	2
DPAR	*Diploneis parma* Cleve	4	2.4
ESOR	*Epithemia sorex* Kutzing	4.4	2.8
EPRA	*Eunotia praerupta* Ehrenberg var. *Praerupta*	4.8	2.3
FCRO	*Fragilaria crotonensis* Kitton	3.6	1.7
FVAU	*Fragilaria vaucheriae* Kützing Pet.	3.6	1.7
FVUL	*Frustulia vulgaris* Thwaites	4.8	2.7
GACU	*Gomphonema acuminatum* Ehrenberg	3.6	1.9
GANG	*Gomphonema angustatum* Kützing	3.6	1.9
GCLA	*Gomphonema clavatum* Ehrenberg	3.6	1.9
GGRA	*Gomphonema gracile* Ehrenberg	3.6	1.9
GPAR	*Gomphonema parvulum* (Kützing) var. *Paravulum*	3.6	1.9
GYAT	*Gyrosigma attenuatum* (Kützing) Rabenhorst	3.9	2.8
GSPE	*Gyrosigma spencerii* Ouekett	3.9	2.8
NACO	*Navicula accomoda* Hustedt	3.4	1.9
NCPR	*Navicula capitatoradiata* Germain	3.4	1.9
NCIN	*Navicula cincta* (Ehrenberg) Ralfs in Pritchard	3.4	1.9
NCRY	*Navicula cryptocephala* Kützing	3.4	1.9
NCUS	*Navicula cuspidata* Kützing	3.4	1.9
NFRU	*Navicula frugalis* Hustedt	3.4	1.9
NGRE	*Navicula gregaria* Dankin	3.4	1.9
NMEN	*Navicula menisculus* Schum.	3.4	1.9
NPUP	*Navicula nyassensis* O. Muller	3.4	1.9
NSHR	*Navicula schroeteri* Meister var. *schroeteri*	3.4	1.9
NCLA	*Nitzschia clausii* Hantzsch	1	2.3
NCMU	*Nitzschia commutatoides* Lange-Bertalot	1	2.3
NDME	*Nitzschia dissipata* (Kützing) Grunow var. *media*	1	2.3
NIFR	*Nitzschia frustulum* (Kützing) Grunow	1	2.3
NLIN	*Nitzschia linearis* W. Smith	1	2.3
NZLT	*Nitzschia linearis* (Agardh) W.M.Smith var. *tenuis*	1	2.3
NPAL	*Nitzschia palea* (Kützing) W. Smith	1	2.3
NSIO	*Nitzschia sigmoidea* (Nitzsch.) W. M. Smith	1	2.3
PFIB	*Peronia fibula* Breb .ex Kützing	5	3
PACO	*Pinnularia acoricola* Hustedt	4.7	2.3
PBIC	*Pinnularia biceps* Gregory	4.7	2.3
PLEP	*Pinnularia leptosoma* (Grunow) Cleve	4.7	2.3
PNOB	*Pinnularia nobilis* Ehrenberg	4.7	2.3
PTRP	*Pinnularia tropica* Hustedt	4.7	2.3
SHAN	*Stephanodiscus hantzschii* Grunow	2.9	1.4
SANG	*Surirella angusta* Kützing	3.6	2.2
SOVI	*Surirella ovalis* Brebisson	3.6	2.2
SULN	*Synedra ulna* (Nitzschia) Ehrenberg	3.1	1.8

Appendix B

Macroinvertebrates taxa identified in the Kebena, Little and Big Akaki rivers, with their habitat preference, feeding group and tolerance values (Bouchard, 2004)

Macroinvertebrate family	Order	Habitat	Tolerance value
Chironomidae (Chir)	Diptera	Lentic	8
Muscidae (Musc)	Diptera	Lentic, Lotic	6
Tabinidae (Tabin)	Diptera	Lentic, Lotic	6
Ceratopogenidae (Cerat)	Diptera	Lentic, Lentic depositional	6
Simuliidae (Simu)	Diptera	Lotic erosional	6
Dolichopodidae (Doli)	Diptera	Lentic depositional, Lentic	4
Culicidae (Culi)	Diptera	Lentic, Lotic depositional	8
Psychodidae (Psyc)	Diptera	Lentic marginal, Lotic marginal	10
Leptophlebiidae (Lept)	Ephemeroptera	Lotic erosional	2
Baetidae (Baet)	Ephemeroptera	Lotic erosional and depositional	4
Culicidae (Culi)	Diptera	Lentic, Lotic depositional	8
Psychodidae (Psyc)	Diptera	Lentic marginal, Lotic marginal	10
Caenidae (Caen)	Ephemeroptera	Lotic and Lentic depositional	7
Elimidae (Elim)	Coleoptera	Lotic erosional	5
Heptageniidae (Hept)	Ephemeroptera	Lotic erosional, Lentic erosional	4
Siphlonuridae (Siph)	Ephemeroptera	Lentic erosional	7
Oligochaeta (Olig)	Oligochaeta	Lentic depositional	8

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