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Ecological impact assessment of hydropower generation in river systems

Thesis submitted in fulfilment of the requirements for the degree of Doctor (PhD) in Applied Biological Sciences

Dutch translation of the title:

Ecologische impact beoordeling van waterkrachtgeneratie in riviersystemen

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List of abbreviations

AIC	Akaike Information Criterion
ANOSIM	ANalysis Of SIMilarities
BMWP-Colombia	Biological Monitoring Working Party-Colombia
BOD	Biochemical Oxygen Demand
CCA	Canonical Correspondence Analysis
COD	Chemical Oxygen Demand
DCA	Detrended Correspondence Analysis
DO	Dissolved Oxygen
DPSIR	Driver-Pressure-State-Impact-Response
EC	Electrical Conductivity
FTU	Formazin Turbidity Unit
GLMs	Generalized Linear Models
NMDS	Non-metric Multidimensional Scaling
NO_3^-	Nitrate
NO_2^-	Nitrite
NH_4^+	Ammonium
oPO_4^{3-}	Orthophosphate
SIMPER	SIMilarity PERcentages
SWOT	Strengths-Weaknesses-Opportunities-Threats
TDS	Total Dissolved Solids
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
TITAN	Threshold Indicator Taxa ANalysis
VIF	Variance Inflation Factor

Chapter 1: General introduction

1.1 Problem definition

Water, energy, food and ecosystems are all essential for human well-being and sustainable development. However, water, food, energy and ecosystems are closely interconnected and highly interdependent (WWAP, 2014). Global projections show that the demand for freshwater, energy and food will increase significantly over the next decades due to different pressures such as population growth, economic development, urbanization, cultural and technological changes, and climate change (Hoff, 2011). The increase in world population and growing wealth are raising the demand for energy and food (Pittock et al., 2016). It was estimated that global energy consumption is projected to grow by up to 50 percent by 2035 (IEA, 2009). Hydropower is a globally recognized source of clean energy, which has played an important role for the international energy supply (UNIDO and ICSHP, 2016). Driven by the increasing demand for energy and global climate change, many countries have given priority to hydropower development for the expansion of their energy sectors (UNIDO and ICSHP, 2016). However, it may also result in increased competition for water between different sectors (e.g water, energy, agriculture, fisheries) and thus damage to ecosystems. Conserving freshwater biodiversity is globally important to ensure ecosystem integrity and sustainability (Jun et al., 2016), and is part of determining an optimal balance between the different uses of water resources.

Dams have made significant contributions to human development and the benefits derived from them have been considerable (World Commission on Dams, 2000). Dams have been built for many reasons such as flood control, irrigation, domestic water supply, navigation, hydropower and recreation (Tullos et al., 2009; Yi et al., 2014). Hydropower is an important renewable energy, which accounted for 16% of the global electricity production in 2011 (Zarfl et al., 2015). In many tropical countries, such as Vietnam, hydropower contributed to 40% of the total electricity production of the country in 2013 (Pham, 2015). It is clear that hydropower is an important energy source for many countries and regions. The most biodiverse river basins around the world, such as the Amazon, Congo and Mekong are now experiencing an unprecedented boom in the construction of hydropower dams (Winemiller et al., 2016). In South America, 2215 hydropower projects which have been planned during the period 2009-2011, will add new dams in 673 undammed and 388 dammed rivers (Kareiva, 2012). In contrast, more than 1100 dams have been removed for river restoration in the United States over the past 30 years (Magilligan et al., 2016). Whilst the increase in the number and frequency of hydropower projects addresses important energy needs, it often overestimates economic benefits and underestimates far-reaching effects on biodiversity and critically important fisheries (Winemiller et al., 2016). Hydropower dams have been the subject of controversy due to their complex social, political and environmental impacts. In this context, one of the key questions is: ‘What is the optimal portfolio of dams for meeting our energy needs while conserving biodiversity for our changing world?’ (Kareiva, 2012). Being aware of the trade-offs between water for hydropower generation, food production, ecosystems, … can inform policy makers to make a plan for hydropower development, licenses issued for new dams and relicensing existing dams.

A typical hydroelectric plant includes three parts: a storage reservoir, an electric plant where the electricity is produced, and a dam that can be opened or closed to control water flow (Fig. 1.1). The reservoir stores a significant volume of water behind the dam. The water then flows by gravity through an intake into a pipe (penstock) which is located inside the dam. At the end of the penstock, there is a turbine propeller, which is turned by the moving water to generate electricity. This same volume of water continues moving by gravity through the tailrace into the river below the dam. The amount of electricity that can be generated depends on a number of factors including the volume of water and its flow rate (U.S Department of the Interior Bureau of Reclamation, 2005).

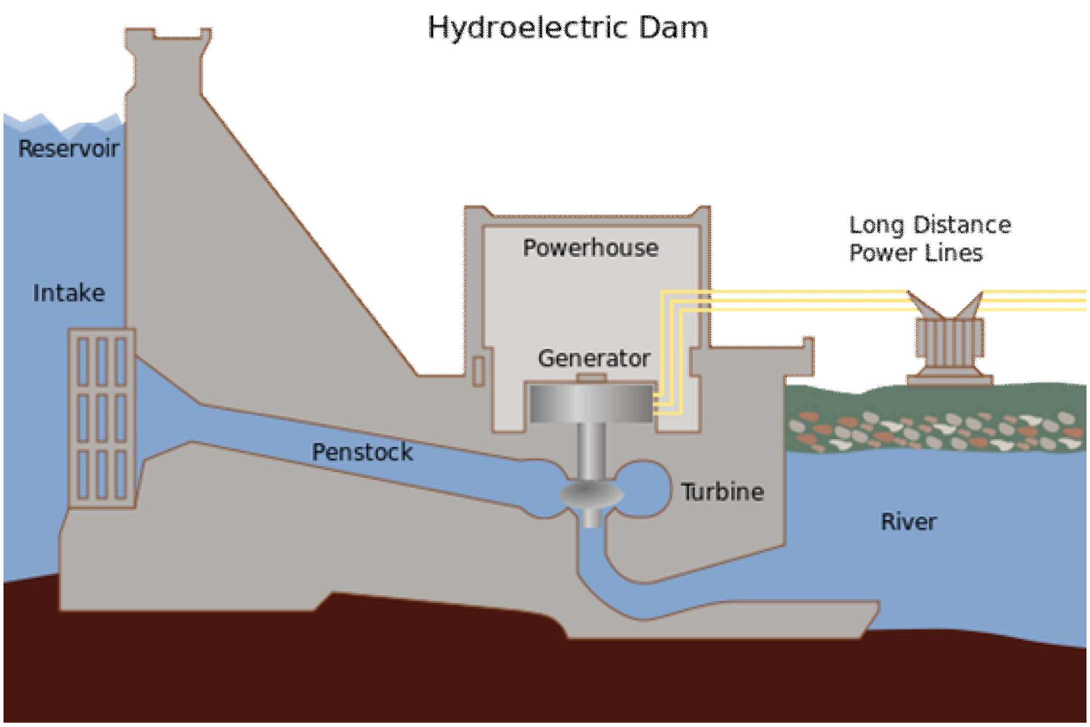

Figure 1.1 A schematic view of a hydroelectric dam (Source: wikimedia.org)

Damming of a river can result in serious negative consequences, including changing physical-chemical parameters (e.g. temperature, dissolved oxygen and suspended solids) and hydro-morphological conditions (e.g. water depth, stream velocity, and substrate). The various specific impacts of dams are described in detail in Chapter 2. In general, a dam stores and restricts water within the reservoir, regulates water flows, prevents or reduces movements of aquatic and terrestrial organisms/animals, and disrupts downstream transport of materials (Castello and Macedo, 2016). A case study from the Mekong river basin has shown that dams will block critical fish migration routes, in that way a dams can have severe impacts on fish productivity and biodiversity (Ziv et al., 2012). In addition, Gerd Marmulla (2001) indicated that the major concern throughout Asia is that dams block the movements of migratory fishes along river courses. However, dams often produce favourable conditions for the development of aquaculture on the reservoir. Therefore, in general, the impact of dams on fisheries/productivity depends on spatial (local areas, whole basin) and temporal scales (e.g. decades, centuries) of interest as well as specific human alteration and exploitations. The proliferation of algal blooms and eutrophication have also been recorded in many tropical and subtropical reservoirs, for example in eight tropical hydroelectric reservoirs in Brazil, Paraguay, and Argentina. These include the Funil reservoir in Brazil (Rangel et al., 2016) and the El Carrizal Reservoir in Argentina (Beamud et al., 2015). Together with eutrophication, there is growing concern for high levels of organic carbon burial in sediment and low oxygen concentrations in the water close to the bottom of the water column (Mendonca et al., 2016) in hydroelectric reservoirs. In addition, depending on its design and other environmental and climactic factors, sedimentation in reservoirs can be a significant problem; it was estimated that the average volume loss in reservoirs due to sediment deposition was 0.93% per year (Luis et al., 2016).

Ecuador has a land area of 256.370 km^2, with a total population of 16.14 million people in 2015 (FAO, 2016). The country is located in South America, which is one of 25 biogeographically distinctive hotspots of biodiversity in the world (Brooks et al., 2002). It has 31 river basins, of which 24 are river basins draining into the Pacific Ocean and seven river basins draining into the Amazon river (Proano, 2005). Up to 2012, Ecuador had 31 hydropower dams (Consejo Nacional de Electricidad, 2013). By 2016, its existing hydroelectric projects produced about 50% of the national electricity (Ecuadorian Rivers Institute, 2016). The economic impact of the hydropower plants in Ecuador is positive (Salazar and Rudnick, 2008). In 2013, the Government of Ecuador announced its decision to invest 7427 million USD to construct 25 new hydro dams in order to meet the country's predicted increase in demand for electricity during the period 2013-2022 (Consejo Nacional de Electricidad, 2013). As a consequence, many rivers throughout the country are and will be further fragmented. Current plans for hydropower dams in Ecuador may result in further degradation of biodiversity and ecosystems. The Guayas and the Portoviejo river basins are important watersheds that have experienced significant anthropogenic changes in the past several decades. The Poza Honda dam is located in the Portoviejo River and started operating in 1971 (U.S. Army Corps of Engineers, 1998). The Daule-Peripa dam started to operate in 1992 and is located in the Daule River in the Guayas River basin (Gelati et al., 2011). Both Daule-Peripa and Poza Honda dams were designed, constructed and installed to fulfill the increasing energy demands of Ecuadors growing population. Since dam projects have first been executed across the country, the formation and development of large carpets of the aquatic macrophyte water hyacinth (*Eichornia crassipes*) on the reservoirs' surfaces has became in many cases a major challenge for sustainable operation of the country's hydroelectric schemes. In Ecuador, environmental problems are particularly severe related to the spread of invasive water hyacinth within the water bodies of hydroelectric reservoirs, resulting in the degradation of water quality in the downstream regions. The construction of hydropower dams on these major rivers is widely viewed as the primary stressor. Unfortunately, detailed information on the causal relationship between specific hydropower dams and impacts on water quality and associated ecosystems remains scarce in many tropical regions. Studies and the evidence base in the peer reviewed or grey literature on the effects of dams are generally lacking in an Ecuadorian context. Therefore, it is necessary to identify the ecological impacts of hydropower dams on Ecuadorian rivers to support management strategies for protecting and restoring Ecuadorian freshwater ecosystems.

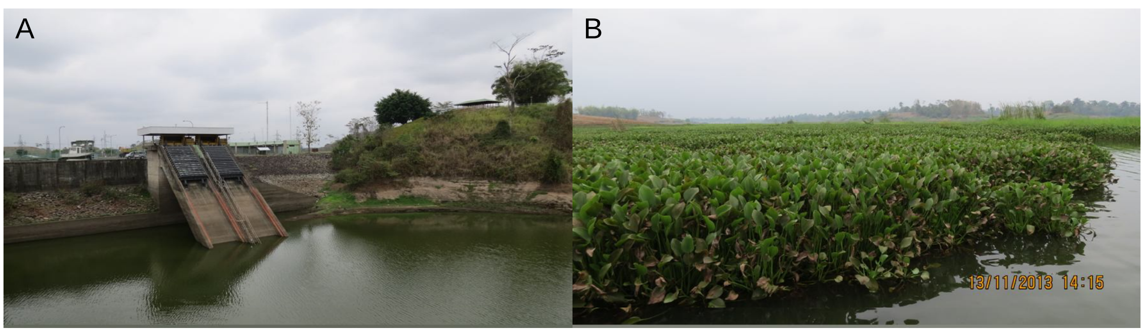

Figure 1.2 Pictures taken at (A) The inlet of the Daule-Peripa dam and (B) Mat of infestion of water hyacinth (*Eichhornia crassipes*) at the Daule-Peripa hydro reservoir

1.2 Scope and objectives

The aim of this study is to analyze the impacts of hydropower dams on water hyacinth and aquatic macroinvertebrates in Ecuadorian rivers. For this study, two river basins have been selected to make this investigation (cf. Chapter 3): the Guayas and Portoviejo river basins. In order to assess the impact of dams on river ecosystems, we first need to know how are the macroinvertebrates distributed in general along the river system. The information about the distribution of macroinvertebrates along the river (gradients), can serve as a basis to clarify the impact of damming and how it can be distinguished from other human disturbances. It was expected that the interrelation between physical-chemical variables, the presence of water hyacinth and macroinvertebrate communities are able to illustrate the ecological impacts of damming and resulting anthropogenic disturbances in Ecuadorian rivers at various spatial scales. The results can serve as the initial steps to determine the ecological impacts of hydropower dams on tropical rivers.

There are four sets of major research questions that this PhD dissertation focuses on:

1. How can models be used to assess the ecological impacts of hydropower dams? (Chapter 2).

2. What are the habitat preferences of macroinvertebrates in a tropical reservoir? Does the development of water hyacinth affect the ecological water quality and macroinvertebrate communities? What is the link to physical habitat created by water hyacinth? (Chapter 4)

3. How does the ecological water quality change along the Guayas River basin? What is the importance of physical-chemical variables in structuring the macroinvertebrate communities and how is this affected by additional stress via the installation of hydropower dams? Which taxa are most affected by the effects of dams (e.g. due to changes in stream velocity)? (Chapter 5)

4. How does the ecological water quality change from upstream to downstream of a dam in the Portoviejo River basin? What are the key environmental factors affecting the macroinvertebrate communities? How are macroinvertebrates distribute along the river system and what is the effect of the dam on this? What is the link between a hydropower dam and the change of ecological water quality? (Chapter 6)

The thesis is structured around seven chapters (Fig 1.3), that link the general objectives with the data collection, data analysis, and ecological assessement related to the two selected specific Ecuadorian river basins.

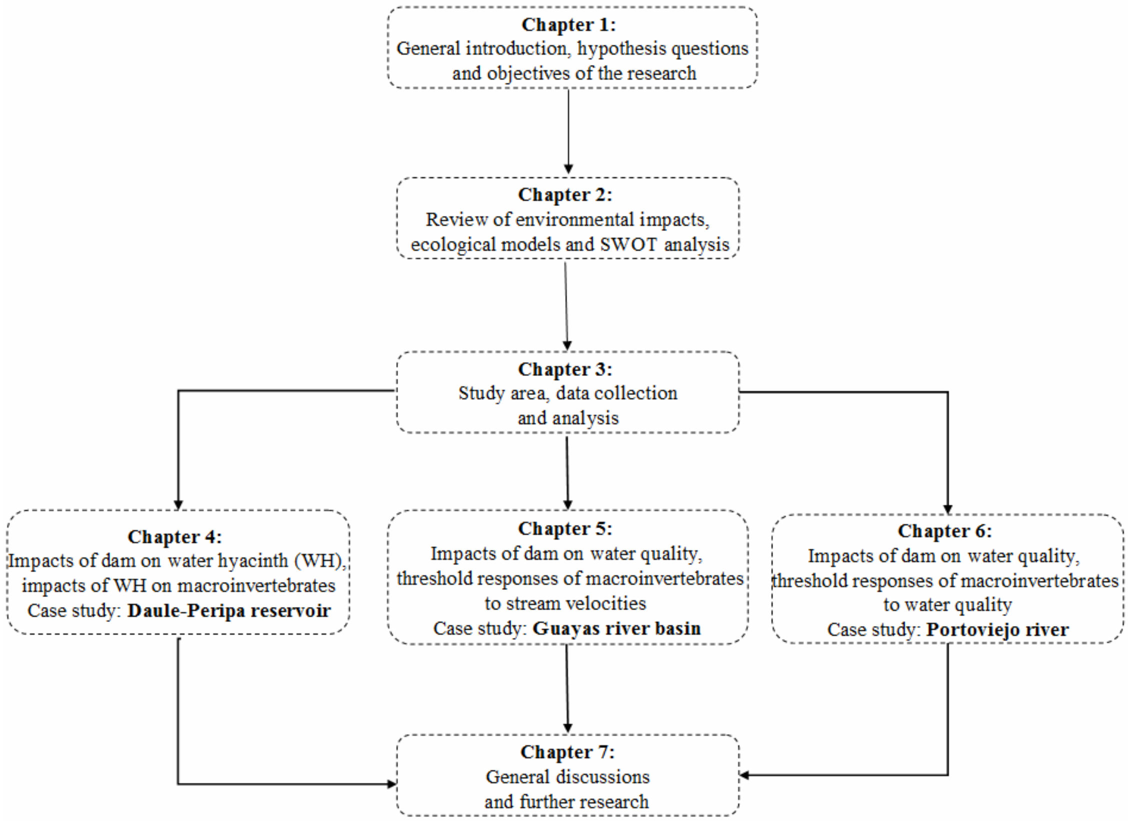

Figure 1.3 Flowchart illustrating organization of research and associated chapters within the thesis

Each of the individual chapters of this dissertation describe the specific goals in order to achieve the overall objectives of this research:

Chapter 1 gives a general introduction to the need for the present study, problem statements, research hypothesis questions and the objectives of the research.

Chapter 2 addresses the first research question above. It presents an overview of the impacts of hydropower dams on river ecosystems. In this chapter, a search for relevant articles was performed using the ISI Web of Science, and the DPSIR framework was applied to review different impacts of hydropower dams on water quality and ecological quality of riverine ecosystems. This chapter also reviews the different ecological models that have been used to assess possible impacts of hydropower dams on water quality and habitat suitability of biological communities. Then followed by the implementation of SWOT analysis to identify and document strengths, weaknesses, potential shortcomings and opportunities for the use of ecological models in impact assessments of hydropower dams.

Chapter 3 deals with materials and methods applied in this dissertation and includes descriptions of the particular study areas. The data were collected in the Guayas River basin in the dry season of 2013 and then in the Portoviejo River basin in the dry season of 2015. Abiotic variables, habitat characteristics, macroinvertebrate community monitoring and analysis methods are described in detail. The chapter also provides a background to applied ecological modeling and assessment techniques.

Chapter 4 addresses the second research question. This chapter is based on our case study of the Daule-Peripa hydropower dam reservoir. We applied Generalized Linear Models (GLMs) in order to disentangle the key elements determining the establishment and spread of invasive water hyacinth (*Eichhornia crassipes*). The relationships between the occurrence of the invasive water hyacinth and water quality properties, as well as macroinvertebrate diversity were examined in order to gain insight into the drivers causing changes in the physical-chemical water quality and diversity of macroinvertebrates in the Daule-Peripa reservoir.

Chapter 5 addresses the third research question. This chapter is based on our a case study of the Guayas River basin. Multivariate analysis was applied to reveal the importance of physical-chemical variables in structuring the macroinvertebrate communities. Since stream velocity is seen as one of the most important environmental factors influencing the distribution of macroinvertebrate communities. Then Threshold Indicator Taxa Analysis (TITAN) was used to examine the threshold responses of macroinvertebrate communities to stream velocity. The chapter also examines which taxa are most affected by changes of stream velocity in the Guayas river basin and which may thus be significantly affected by the construction of hydropower dams.

Chapter 6 addresses the fourth research question and is based on our case study of the Portoviejo River basin. We assessed the physico-chemical and ecological water quality of the Portoviejo River basin. The importance of several environmental factors and habitat characteristics were linked to the ecological water quality. TITAN analysis was again applied to detect the responses of macroinvertebrate communities to changes in key environmental variables and to find potential indicator taxa of macroinvertebrate communities in the Portoviejo river basin.

Chapter 7 then brings together the previous research and findings to then discuss the research tools utilized in data collection, application of ecological modeling, ecological assessment based on macroinvertebrates in hydropower dam impact assessment and practical issues in river management. Following on from this, the thesis concludes with recommendations for further follow on research on hydropower dam impacts which builds on the findings of this study.

This PhD thesis presents case studies in Ecuador. However, the methodology and the results described here may be applicable to other watersheds in (tropical) countries. The information presented here will be particularly useful for management of similar rivers in South-America, as well as the rest of the world facing similar situations.

Chapter 2: Impact of hydropower dams on river ecosystems and the use of ecological models – A review

Based on:

Thi Hanh Tien Nguyen, Gert Everaert, Pieter Boets, Elina Bennetsen, Martin Volk, Thu Huong Thi Hoang, Peter L.M. Goethals. Impact of hydropower dams on river ecosystems: A review from a modeller's perspective. Ecological Engineering (in preparation).

Chapter 2: Impact of hydropower dams on river ecosystems and the use of ecological models – A review

Abstract

We critically analyze a set of ecological models that are used to assess the impact of hydropower dams on water quality and habitat suitability for biological communities. Based on a literature search, we found that the most important variables contributing to structural and functional ecosystem changes are variations in water flow and water depth coupled with increased nutrient availability. Another result is that ecological models can make considerable contribution to the impact assessment of hydropower dams, as via simulations of different scenario's (e.g. with and without dam, different operation methods, …) the related ecosystem shifts can be predicted and analysed. However, one of the remaining shortcomings of these models is the limited capacity to separate dam-related impacts from other anthropogenic influences (e.g. agriculture, urbanization). Moreover, collecting sufficient high quality data to increase the statistical power remains a challenge, as the severely altered conditions (e.g. generation of very deep lakes) also leads to difficulties for standardized data collection. Future opportunities can be found in integrating models to improve the understanding of the different processes affected by hydropower dam development and operation and the use of remote sensing methods for data collection.

2.1 Introduction

Human population growth and economic development increased demands for energy. Hydropower is a renewable and cheap source of energy (Bratrich et al., 2004; Castelletti et al., 2008; Jager and Smith, 2008) which contributes to about 16% of the global electricity production in 2011 (Zarfl et al., 2015). Dams have been continuously constructed for hydropower generation and other purposes (World Commission on Dams, 2000; Alison Bartle, 2002; Kumar et al., 2011). Until 2016, globally, there are about 58,402 large dams (dam with the height higher than 15 m) of which 9,595 dams primarily designed for hydropower generation are in operation (International Commission on Large Dams, 2016). Hydropower dams are typically operated with the goal of maximizing energy benefit, whilst meeting other legal water requirements such as environmental flow to protect aquatic ecosystems (Jager and Smith, 2008). However, besides these benefits, hydropower dams can have negative environmental impacts on the structure and functioning of aquatic ecosystems (World Commission on Dams, 2000; Bunn and Arthington, 2002; Kumar et al., 2011).

Dam construction is considered one of the major pressures contributing to the modification of natural river ecosystems (Tonkin et al., 2009; Zhai et al., 2010). Scientists and engineers have developed numerous methods for mitigating the environmental impact of dams, such as installation of pool-type fish passes, and nature-like bypass channels to facilitate movement of fish from below the dam to the reservoir and further upstream. Specially designed fish elevators that collect the fish in boxes and then lift them to the level of the impoundment (Berkamp et al., 2000). However, not all impacts can be reduced through implementation of a variety of mitigating measures (Berkamp et al., 2000). The impacts of dams to freshwater ecosystems can be devastating and difficult to restore (Schelle et al., 2004). Understanding the trade-offs between water for the environment and water for hydropower in regulated rivers can inform decision making about hydropower system planning, policy and operations (Rheinheimer et al., 2013). Therefore, the ecological impact assessment of hydropower dams on river ecosystems before construction, during operational phases and after dam removal, is indispensable.

Ecological models can support decision making in environmental and conservation management (Schmolke et al., 2010) and have already been applied to assess the impact of dams on fish (Hatten and Parsley, 2009; Hatten et al., 2009; Garcia et al., 2011; Costa et al., 2012; Fjeldstad et al., 2012; Ziv et al., 2012), macroinvertebrates (Marchant and Hehir, 2002; Molozzi et al., 2012; Wang et al., 2013), amphibians (Yarnell et al., 2012) and vegetation (Benjankar et al., 2011; Egger et al., 2012; Guarino et al., 2012). Models have become indispensable in environmental assessment, planning and management (Crout et al., 2008). However, the type of model implemented, the variables included and the parameterization used, strongly differ between different studies. For instance, Freeman et al. (2001) used depth, substrate and flow characteristics as input variables in order to assess the impact of altered flow regimes on the habitat suitability of juvenile fish. Kunz (2011) used dissolved oxygen and nutrient compounds as input variables to develop a water quality model taking into account the impact of a tropical reservoir. Biological communities reflect watershed conditions since they are sensitive to changes in a wide array of environmental factors (Karr, 1981). The biological communities that are exposed to pollutants act as integrators of the multiple present and past environmental effects (Cranston et al., 1996). Moreover, different aquatic communities have different preferences in physical-chemical and morphological conditions. Hence, it is expected that changes in the aquatic community reflect the impact of the dam on the entire ecosystem.

The qualification of models depends on both the objective of the research and the applicability of the model (Guisan and Zimmermann, 2000). Indeed, integration of physical and biological processes helps to select the appropriate input variables and modelling techniques in order to assess the impacts of hydropower dams on river ecosystems and provides effective management strategies (Imhol et al., 1996). In addition, applying the right procedure for model selection, determining the range of application, choosing proper input variables and selecting appropriate evaluation methods are also crucial to obtain a useful model (Guisan and Zimmermann, 2000). Consequently, an overview and critical analysis of the development and use of ecological models to assess the impact of hydropower dams is needed and currently not available. This chapter reviews studies on impacts of hydropower dams on water quality and ecological quality of riverine ecosystems. We begin by describing the complex interactions involved in the potential impacts of hydropower dams based on the DPSIR (Driver-Pressure-State-Impact-Response) framework. The next section is a review of ecological models approaches used to assess possible impacts of hydropower dams. Finally, we investigate the limitations and opportunities for the use of ecological models in hydropower dam impact assessment.

2.2 Methodology

The European Environment Agency's DPSIR (Driver-Pressure-State-Impact-Response) framework was introduced as general guideline to describe cause effect relationship between human activities and environment (Smeets and Weterings, 1999). Due to the framework, there is a chain of causal links between 'Drivers' (social, demographic and economic developments) and 'Pressures' on the environment. The 'Pressures' causes change to the current or future 'State' of the environment. These environmental changes lead to positive or negative 'Impacts', and the "Response" is the action taken to solve potential environmental problems. It can be seen from literature, the DPSIR framework was already applied on water resources management, river basin management (Arias-Hidalgo, 2012; Bell, 2012; Sekovski et al., 2012; Vermaat et al., 2012), river restoration (Song and Frostell, 2012) and biodiversity conservation (Spangenberg et al., 2009). In this study, we studied the available relevant literature regarding the specific issue of hydropower dams and its impacts on river ecosystems. Afterward, the (DPSIR) concept was developed based on our own experience in order to disentangle the complex interactions involved in the impact assessment of hydropower dams on aquatic ecosystems.

As inferred in the DPSIR framework, three kinds of impacts could be identified, characterized by morphological, physical-chemical and biological. In order to provide an overview of the different model approaches were used to assess hydropower dams impacts on river ecosystem, a search for articles was performed in the ISI Web of Science (January 22, 2016, http://apps.webofknowledge.com). Articles were derived from a search in ISI Web of Science using search topic = 'hydropower dam*' AND topic = 'habitat*'. A sub search was performed using topic = 'model*' AND document type = article. Fifty-six key articles were extracted, but those dealing with terrestrial ecology and in which insufficient detailed information on the model development process was given, were discarded. Finally, thirty-two articles were retained. In this chapter we focus on four major questions (1) what types of modelling approaches have been applied for ecological impact assessment, (2) which input variables were used, (3) how models were validated and (4) how models can be applied? Based on 32 reviewed articles, we listed all main input variables that were used; the type of model approached; model validation process and the output of the model. Afterwards, we developed an integrated conceptual model that illustrates the linkages between the main input variables, model approaches, the output variables and biotic, abiotic interactions in the ecosystems related to hydropower dams.

Different modeling approaches have been used to assess the impact of hydropower dams on the watery ecosystem. To further explore the potential of ecological models in hydropower dam impact assessment, an analysis in Strengths, Weaknesses, Opportunities and Threats (SWOT) was implemented. This assessment provides an overview of the advantages and limitations of models which have been developed and identifies the challenges and opportunities for future models in hydropower dam impact assessment.

2.3 Results and discussion

2.3.1 Impacts of hydropower dams on aquatic ecosystems

Figure 2.1 describes the DPSIR framework related to the links between socio-economic drivers, pressures, changing state, impacts on river systems and potential actions can be taken to minimize the ecological problems cause by hydropower dams.

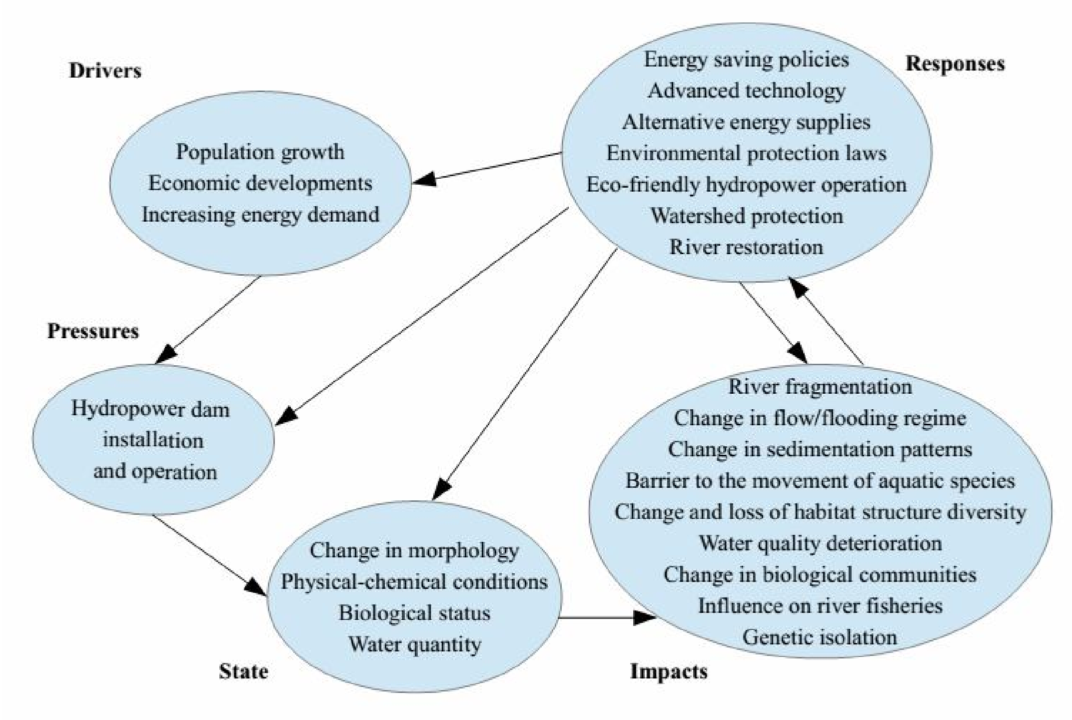

Figure 2.1 Drivers-Pressures-State-Impacts-Responses framework presenting the impact of hydropower dams on the ecological water quality and habitat suitability for aquatic biological communities.

a. Drivers

Rapid growth of the human population, economic development, climate change, and the need to close the electricity access gap have stimulated the search for renewable energy such as wind, waves, tides, biomass, biofuels, and hydropower (Zarfl et al., 2015). Hydropower was considered as a cheap and renewable-energy source (Kumar et al., 2011) because it uses the water from river to generate electricity and it produces lower amounts of greenhouse gases compared with hydrocarbon fueled power generation (Boavida et al., 2015). The (global) increase in energy demand is the driver that triggers the construction of hydropower dams. Especially, future hydropower development is expected to be primarily concentrated in developing countries and emerging economies of Southeast Asia, South America, and Africa (Zarfl et al., 2015).

b. Pressures

Despite their economic benefits, the civil works during the implementation period of the dam project such as the road, power house and inlet construction impact the surrounding river ecosystem (Bruno and Siviglia, 2012). Moreover, once installed and in operation, dams interrupt the natural river continuum (Graf, 2006) and this can lead to changes to the natural flow regime and changes in hydraulic conditions in general (Meile et al., 2011).

c. State

Dam construction and operation are pressures that can cause changes in the state of river geomorphology and hydrology (Vörösmarty et al., 2003) and river isolation (World Commission on Dams, 2000). Furthermore, the construction and presence of hydropower dams can affect the physical-chemical and morphological conditions in a river and thus alter the habitat suitability for riverine communities (Grand et al., 2006; Garcia et al., 2011). Based on a report of the European Water Framework Directive, hydropeaking is one of the main stressors on aquatic ecosystems (EU, 2000). It can dramatically induce river modifications such as alterations to stream banks and channel morphology, water depth, wetted area, velocity distribution, substrate composition, suspended matter, temperature, habitat structure and heterogeneity (Scruton et al., 2008). The damming of river causes changes in sediment transport (Käiro et al., 2011), water depth, water temperature and stream velocity (Ruetz and Jennings, 2000; Sinokrot and Gulliver, 2000; Krause et al., 2005; Hatten and Batt, 2010). Moreover, in some cases huge amounts of sediment that are highly contaminated with heavy mentals, organic matter and pesticides can be deposited in the reservoir (Jacoub and Westrich, 2006; Couillard et al., 2008). The reservoir itself may undergo effects of eutrophication processes, such as algae blooms and floating plant accumulations (Chapman, 1996). The increase of water residence time, eutrophication and decomposition of organic matter can moreover cause the increase of toxicants present upstream of a dam. The toxicants may be transported downstream by future floods and intensive rainfall events. In addition, water level variation due to power generation can propagate and create changes in the tidal regime for very long distances (Jay et al., 2015).

In particular in combination with the discharge of organic matter (e.g. wastewater from urbanized areas) and nutrients (e.g. from intensified agricultural activities) an intensified status alteration can occur, resulting in substantial declines in dissolved oxygen, increased algae growth, substrate modifications, sediment and mud accumulation, toxic algae blooms, etc. In other words, several alterations in the hydromorphological and chemical status, can eventually further boil down to biological alterations.

Organisms are physiologically, anatomically, morphologically and behaviorally adapted for survival in a specific habitat. Thus, the creation or destruction of such habitats can either lead to the rejection or multiplication of related species (Hansen et al., 2005). Modifications of physical-chemical water quality may deteriorate spawning grounds and nursery areas (Yi et al., 2010), lead to biodiversity loss (Chapman, 1996), alter species interaction (Allesina and Tang, 2012) and ecosystem processes (Chapin III et al., 2000). Dams block migration routes for migratory fish (Han et al., 2008; Makrakis et al., 2012; Brown et al., 2013) isolating populations and increasing their risk of extinction (Navarro et al., 2007). Dam construction and operation have also shown to have a significant impact on microzooplankton richness (Zhou et al., 2009) and density and composition of macroinvertebrate communities (Tonkin et al., 2009; Wang et al., 2013). In addition, dams can alter periphyton biomass (Tonkin et al., 2009), provoke changed phytoplankton densities (Zhou et al., 2009) and cause algal blooms (Lessard et al., 2013). Phytoplankton, zooplankton and invertebrates are important food sources for fish and other vertebrates. The change in food availability affects fish growth and survival (Grand et al., 2006) and reproduction of key species (Yarnell et al., 2012). Moreover, hydropower dams change the taxonomic and trophic structure of fish communities and can cause a reduction of fish species richness (Cerny et al., 2003; de Merona et al., 2005) and abundance (Dauble et al., 2003; Yi et al., 2010).

d. Impacts

Damming of rivers is used for multiple purposes, such as agricultural irrigation, flood control, hydropower generation and recreation, so not all environmental impacts associated with dams can be directly attributed to hydroelectric power. Hydroelectric facilities can have a major impact on aquatic ecosystems. For example, fish and other organisms can be injured and killed by turbine blades. However, assessment of the environmental impacts of a specific hydropower facility requires case-by-case review (EPA, 2013) to determine the affected ecosystem services and related socio-economic consequences of the status changes.

e. Responses

The response is all actions necessary to avoid or control the potential adverse environmental and socio-economic impacts concerning hydropower projects and enhance environmental opportunities. The societal responses to the corresponding drivers, pressures, state of the environment or impacts via various prevention, mitigation, or adaptation measures regarding to the environmental problems (Smeets and Weterings, 1999).

The response related to drivers for dam construction is energy demand management. There are many potential management options could take to minimize the energy demand. For example, many European countries (e.g. Italy and United Kingdom) are applying the energy-saving policies such as interruptible tariffs or time of day pricing (Torriti et al., 2010). Another option can be promoting for the use of advanced technology (e.g. energy-saving building and appliances) in order to reduce energy consumption (Omer, 2008). In addition, the government should apply the policies to enthuse people to use alternative renewable-energy resources such as solar power, wind power and biomass (Islam et al., 2006).

To reduce the pressure of hydropower dams, the management options could be the development and application of environmental laws to hydropower projects. For example, in United States, regulators issue licenses for a limited term of 30 to 50 years. After the license expires, the operator must obtain a new license compliant with current environmental laws. During relicensing, the Federal Energy Regulatory Commission evaluates the performance of the dam project and to determine if the dam can continue to operate (Federal Energy Regulatory Commission, 2016). Literature shows that the impact of dams depends on its location, design and operation (Dugan, 2008). Comparing to the large dams built for irrigation, water supply and flood control, dams built for hydropower have separate objectives, involve distinctive components, respond to unlike markets and are operated in different ways (World Commission on Dams, 2000). Therefore, alternatives to classic hydropower dams such as run-of-the-river and pumped-storage minimize impacts related to damming construction (Sternberg, 2010). If the dam with storage reservoir needs to be constructed, it is important to consider the location of the dam, because the dam expansion in regions with high endemism species would become more harm to imperiled aquatic resources (McDonald et al., 2012). The dam design is also important, as fish passage highly depends on dam design (Schilt, 2007) or the construction of fish passage facilities (Yin et al., 2012).

Although being a renewable electricity source, hydropower is also accompanied by significant environmental impacts on river ecosystems which are an (in)direct cost to society (McCartney et al., 2001). Hence, to maintain and/or restore the initial state of the environment we need to control and minimize the impacts (Smeets and Weterings, 1999; Maxim et al., 2009). The ecologically sound dam management which mimicking the natural environmental flows (Richter and Thomas, 2007), changes in release patterns and operation schemes (Grill et al., 2014) which aim to improve environmental flows and ecosystem services. For the dams with considerable storage reservoir, the hydropower generation interruptions could be replaced with temporal increases of flow release in order to offers more protection to the most vulnerable stream sections while maintaining the natural flow paradigm in rivers affected by hydropeaking (Vezza et al., 2014). In addition, the impacts can be mitigated by river restoration actions such as removing barriers to migration and restocking rivers.

2.3.2 Models for hydropower dam impacts assessment

Table 2.1 provides a synopsis of the ecological modeling approaches that have been developed to assess the impact of hydropower dams on aquatic habitat suitability. The models were reviewed per aquatic community based on modelling approaches, input variables (geomorphology, hydrology, meteorology, physico-chemistry and biology) and model validation (Table 2.1).

Table 2.1 Summary of reviewed articles that used models as a tool to assess the impact of hydropower dams on habitat suitability.

	Total	Hydrodynamic Model	Water Quality Model	Habitat Suitability Model	Integrated model
Input variables					
Geomorphology					
Latitude/longitude	**2**		1		1
Elevation	**6**	2	1		3
Stream gradient	**4**			2	2
Sinuosity	**1**			1	
River Width	**4**			3	1
Depth	**16**			9	7
Substrate	**13**			6	7
Substrate roughness	**2**	1			1
Hydrological					
Velocity	**16**			9	7
Discharge/flow	**19**	3	2	4	10
Hydrological regime	**1**			1	
Meteorological					
Air temperature	**5**		2		3
Air pressure	**1**		1		
Cloud Cover	**3**		2		1
Wind Speed	**3**		2		1
Humidity	**2**		1		1
Rainfall	**1**	1			
Solar radiation	**2**		1		1
Physical-chemical					
Water temperature	**11**		3	4	4
Total dissolved solid	**1**			1	
Toxicants	**1**	1			
Suspended sediment	**3**	1		1	1
DO	**4**		1	2	1
Conductivity	**2**			2	
pH	**2**			1	1
Turbidity	**2**			1	1
Nutrient	**2**		1	1	
Chloride	**1**			1	
Oxidation reduction potential	**1**			1	
Calcium ion	**1**			1	
Salinity	**1**			1	
Ligneous structure	**1**			1	
Biological components					
Fish	**16**		1	9	6

Macroinvertebrates	**3**			2	1
Amphibians	**1**	1			
Algae	**1**			1	
Model validation					
Cross validation	**4**	1	1	1	1
Single validation	**12**	2	1	5	4

2.3.2.1 Models approached and practical application

Hydropower dam assessment models can be divided in four categories: hydrodynamic models, water quality models, habitat suitability models and integrated models. Integrated models combine two or more models (Fig. 2.2). Among the reviewed articles, four papers used hydrodynamic models, and three authors used water quality models to assess the impact of hydropower dams on the river morphology and on the physical-chemical quality of the river. Sinokrot and Gulliver (2000) used a water quality model to determine the impact of flow variation caused by hydropower dam operation on the river water temperature. A habitat suitability model was used in 12 papers to evaluate the impact of the changes in physical-chemical conditions on the habitat of biotic communities. Li et al. (2011b) used water depth, velocity and water temperature as input variables in their habitat suitability model to calculate the minimum flow for fish habitat conservation. The majority of articles (13) used an integrated model, in which the impact of hydropower dam construction on river morphology was linked to water quality, food web ecology or biological preferences (Fig. 2.2). There was no individual food web model; however, in one paper it was considered as a part of an integrated model.

Models were also used to assess the success of restoration and mitigation actions on habitat suitability of fish (e.g. Bartholow et al., 2004; Fullerton et al., 2009) and macroinvertebrates (e.g. Gore and Hamilton, 1996). As can be seen by the study of Bartholow et al. (2004) involved the application of a one-dimensional hydrodynamic model coupled with a water temperature model. They used the model to assess if the growth and survival of brown trout could be improved via thermal habitat enhancement by providing an understanding of flow effects on temperature. Habitat suitability models and integrated models were mainly developed to support decision making related to the management of multifunctional dams (e.g. hydropower generation, water supply) and fish habitat conservation related to flow management (e.g. minimal flow requirements). Owen et al. (1997) used stream flow to construct fuzzy membership functions in order to model variability in experts' perceptions associated with reservoir operation for hydropower generation, fish habitat, recreation (kayaking) and scenery preservation.

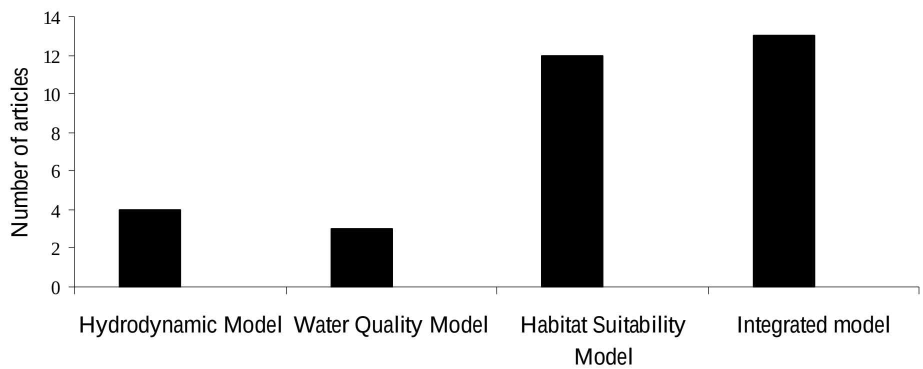

Figure 2.2 The uses of models in hydropower dam impact assessment (based on the 32 articles that were retained).

2.3.2.2 Input variables

There is a wide spectrum of environmental variables that can change over time and space (Ruokolainen et al., 2009). The complex task of dam impact assessment needs careful selection, interpretation and weighting of a multitude of biotic and abiotic information by means of models and expert knowledge (Grill et al., 2014). Therefore, before developing the models, it is necessary to select appropriate input variables (Guisan and Zimmermann, 2000). It has been highlighted that the model outcome depends on the assumptions made in the pre-processing steps of the modelling process (Zuur et al., 2010). Thus, the choice of appropriate variables is a crucial step in model development (Everaert et al., 2012).

Overall, 32 different input variables were used, which can be classified as (1) geomorphological, (2) hydrological, (3) meteorological and (4) physical-chemical input variables (Table 2.1). The number of input variables used per study ranged from one to 15, but usually three to six variables were retained in the model. Mainly morphological (e.g. depth, substrate), hydrological (e.g. flow, velocity) and physical-chemical variables (e.g dissolved oxygen) were implemented. Nineteen reviewed articles used flow (discharge) and 16 papers used hydro-morphological variables related to flow (e.g. depth, velocity) as input variables. Eleven papers used water temperature, and 13 papers used substrate and less than five papers used meteorological data as input variables. Many hydro-morphological data such as depth, substrate, river flow, velocity and dissolved are commonly easy to collect and moreover often freely available in databases. Water depth and current velocity are important factors for spawning habitat (Rosenfeld et al., 2011). Moreover, the velocity/depth ratio may be the best determinants of habitat type and, which is the most distinct among habitat types (Jowett, 1993). However, pH, BOD, metal concentration, nutrient concentrations are key factors affecting the presence of aquatic organism (e.g. macroinvertebrates) but seemed in most cases neglected in ecological impacts assessment. Therefore, those variables should be monitored and included into the models (Hoang et al., 2010) as they can contribute to a better assessment on ecosystem alterations, but also the social-economical consequences, such as increased needs for water purification.

2.3.2.3 Model processes and outputs

The models have been used to assess an expanding range of impacts of hydropower dam at various spatial and temporal scales. Existing ecological models provide a strong basis to assess the impact of changing hydrological regimes and water quality on the habitat suitability of fish, macroinvertebrates and algae.

Flow is the key driver in hydropower dam modeling processes (Fig. 2.3). Flow is used as the input variable for hydrodynamic and water quality models to predict the physical and chemical impacts such as change in water depth, velocity and other variables driven by flow regulations. In some cases, outputs from water quality or hydrodynamic models have been used as the input variables for habitat suitability models to predict the impact of dams on habitat suitability. For example, Krause et al.

(2005) used models to assess the impact of flow manipulation by hydropower on water temperature. In a next step, water temperature was used as the input variable to assess the habitat suitability of fish. On the other hand, flow has also been used as the input variable in habitat suitability models to predict the influences of flow management on habitat suitability of fish (Enders et al., 2009) or macroinvertebrates (Li et al., 2009). In addition, the biomass of invertebrates is used as an input variable in food web models to predict the changes of vertebrate communities (Grand et al., 2006).

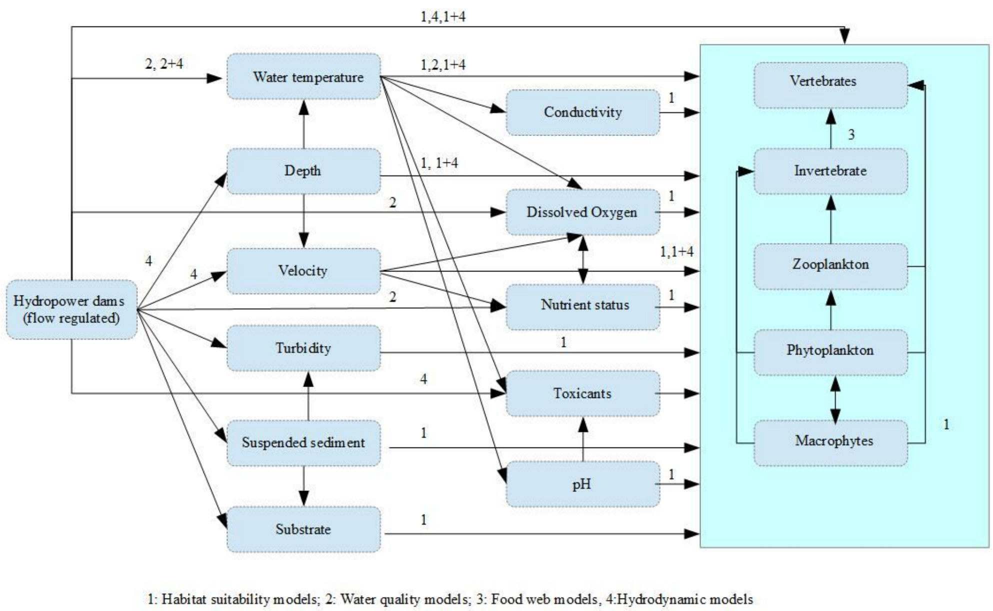

Figure 2.3. Overview of main input variables used for the models, and a representation of the biotic and abiotic interactions related to hydropower dams. The arrows point from an input variable to an output variable. The number indicates the type of model implemented: habitat suitability model (1); water quality model (2); food web model (3) and hydrodynamic model (4); (+) Indicates the integrated approach.

2.3.2.4 Model validation

Validation (model evaluation) is a testing process to check whether the model is acceptable for the intended purpose (Rykiel, 1996). As such, model validation is an indispensable step prior to model acceptance (Mayer and Butler, 1993). A good way to validate model is a comparison of simulated data with observed data of the real system. If the output of a model corresponds with observed data, then the model is an adequate representation of the system. In case of lacking field data, model validity can be supported by expert insights on model behavior and results. Ecological models are often built for understanding (e.g. scientific research, practical management) and prediction (forecasting) purposes. However, an ecological model conceptualizes real life (Guisan and Zimmermann, 2000) as it is not feasible to integrate all assumptions. In the reviewed papers, models were validated using an independent test set (12 papers) or a cross-validation procedure (four papers). In sixteen papers, no model validation procedure was reported. For example, Quiroga et al (2015) developed a model to predict the losses of fish habitat by the construction of two larger hydropower dams, there were thus no data available for model validation. Classic model validation included model comparison between predictions and observations from an independent dataset (Cerny et al., 2003). Alternatively, models were validated by comparing results with field observations from a similar ecosystem (Gore and Hamilton, 1996; Krause et al., 2005).

2.3.3 Strengths-Weaknesses-Opportunities-Threats (SWOT) analysis

Various models have been developed to get insight into the impact caused by hydropower dam construction and operation. However, within the range of approaches that have been implemented and described in literature, no best practice for model development, validation and use could be found. Model selection in the reviewed papers was based on the preference of the model developers and on objective parameters. Often site-specific criteria decide which approach is most suitable. In order to select the right model, a holistic approach is needed, considering the interplay between many elements such as the purpose of the model, the type of present data, the available knowledge (Thuiller et al., 2010) and the model outputs that are required (Schmolke et al., 2010). Hence, the selection of an appropriate model should not only depend on statistical considerations (Guisan and Zimmermann, 2000; Everaert et al., 2013; Boets et al., 2014). In general, good modelling practices are i) clear purpose, ii) adequate assumptions, implications and reporting results, and iii) serious evaluation (Crout et al., 2008). Modelling results should be easy to present to different stakeholders and should be widely applicable (Conallin et al., 2010). We used a strengths-weaknesses-opportunities-threats (SWOT) analysis to discuss challenges and opportunities of applying ecological models in hydropower dam assessment (Table 2.2).

Table 2.2 SWOT analysis of models used in hydropower dam impact assessment

Strengths	**Weaknesses**
-Suitability for exploring various types of dam impacts - Enabled syntheses of expert knowledge and empirical data -Applicable at different scales	-Ignore the interactions among physical-chemical properties and biological responses -Simplicity in variable selection -Lack of clear statistical criteria to assess the models and related thresholds
Opportunities	**Threats**
-Increasing environmental data quality and availability -Growing interest and technical advances in ecological modeling -Integrated models become better and more reliable	- Over or under prediction - Expenses for data collection

a. Strengths

The first strength of using models in hydropower dam impact assessment is the potential to explore the change of water quality and habitat suitability under different dam management strategies (scenario-based analysis). Kunz et al. (2011) found for instance that daily discharge data could be used to evaluate the effect of future changes in riverine sediment and nutrient concentrations on water quality. Hatten and Batt (2010) used depth and velocity simulations to predict the distribution of fish under different management scenarios and selected the most successful restoration actions. To explore the potential effects of flow variation related to hydropower on amphibians, Yarnell et al. (2012) used two-dimensional hydrodynamic modeling to simulate how hydraulic conditions vary between distinctive flow scenarios. In a next step, the output from the flow simulation was used as input variable for a habitat suitability model to quantify biodiversity loss under several pulsed flow scenarios. Model simulations can detect specific negative impacts on the aquatic biota, allowing restoration strategies to be focused on especially threatened species (Vezza et al., 2014). Habitat modeling can be a suitable tool to explore the impacts of hydropeaking flows on habitat availability, thus improve the understanding of long-term effects of hydropeaking on different life-stages of organisms and their abundance (Boavida et al., 2015).

Models can be used to quantify the ecosystems' disturbance level compared to a given reference state (Boavida et al., 2015). They can be used for generating baseline data (e.g. simulation of conditions without dam) and predicting the habitat losses or habitat recovery. Predictive tools can be used to forecast future changes due to dam installation or dam removal. In addition, it can identify which restoration strategy is best under a variety of alternative management actions. As a consequence, predictive tools allow managers to assess potential outcomes before making costly decisions (Fullerton et al., 2009).

The second strength is the possibility to integrate data originating from different monitoring campaigns and knowledge from literature and experts. For instance, meteorological information is easily obtained from meteorologic stations whereas river characteristics such as depth, substrate composition and water temperature can be measured in the field using a standardized sampling protocol with relatively low cost. In addition, models can assist in reducing amount of data. Data collection is often pose difficulties in sampling in large rivers, remote locations and extreme high flow conditions (Quiroga et al., 2015). In cases where data are lacking or no data are available (e.g. inaccessible location, no pre-impact data available), knowledge-based models are useful tools (Ahmadi-Nedushan et al., 2006; Jørgensen, 2008) because, information can be obtained from literature or from local and academic expert knowledge. Boavida et al (2015) developed a fuzzy logic model to investigate the effects of hydropeaking in the habitat of fish in the Ocreza River, Portugal. This synthesizing expert knowledge and river engineering allows the use of simulated environmental conditions (e.g. depth, velocity) as predictors. Additionally, data-driven models can integrate with expert knowledge-based approaches and hence improve model reliability (Mouton et al., 2009). A last strength of most models is that they can be applied at a wide range of spatial scales varying from small creeks (Grand et al., 2006; Cioffi and Gallerano, 2012) to whole river basins (Zhai et al., 2010) or for different time scales (Grand et al., 2006; Yarnell et al., 2012).

b. Weaknesses

A notable weakness of existing ecological models used in hydropower dam impact assessment is that they are mostly not able to simulate (accurately) interactions among physical-chemical properties and biological responses. In the ecosystem, there are complex interactions between organisms and their biotic and abiotic environments (Anand et al., 2010). Therefore, ecological models must be constructed based on the good understanding of the reactions and the processes of ecosystems (Jorgensen et al., 2009). However, the models presented in the selected papers often disregard the complex processes of ecosystems (Cioffi and Gallerano, 2012). Although a few models combined two or more model types, no integrated protocol for model development has been proposed.

Understanding which factors are important, as well as being able to describe the interaction between factors will help managers to determine the potential management strategies (Krause et al., 2005). For instance, flow alteration causes changes in water temperature (Sinokrot and Gulliver, 2000; Krause et al., 2005) and this change in temperature also affects other variables such as pH and dissolved oxygen. These relational shifts can influence the survival, growth and reproduction of aquatic species (Yarnell et al., 2012). Most of the models reviewed in this paper have not taken important ecological control mechanisms into consideration (such as nutrient competition or the effect of zooplankton feeding on phytoplankton concentrations). Hydrodynamic models and water quality models only predict the changes of the physical-chemical water quality of the river, but often lack biological components. Several water quality models considered the impact of discharge on water temperature, but did not elucidate on how temperature changes influence aquatic biota or how fluctuations in flow determined the habitat of aquatic biota.

Most papers often just represented one piece of the complicated processes related to hydropower dam impact. For example, Sinokrot and Gulliver (2000) studied flow impact on river water temperatures, but they did not address the impact of temperature changes on aquatic habitat or the impact of flow fluctuations on aquatic habitat issues. Although Yarnell et al. (2012) determined habitat suitability based on the abiotic or biotic conditions for a single target species or life stage, their models were not able to analyze the interaction of environmental variables on the survival, growth and successful reproduction of a particular species. Moreover, Null et al.

(2014) used estimates of suitable fish habitat for an entire river; however, habitat segments were not all connected so model prediction may become overestimated of habitat suitability for anadromous fish or other migratory species. Habitat suitability models can be used to predict the availability of habitats appropriate for a species occupation under varying flow conditions and to assess what flows may potentially limit or enhance reproductive output (Bondi et al., 2013). Null et al. (2014) estimated fish habitat which is linked to fish population dynamic, but it is not the good substitute. Models are not be able to predict future densities of populations because the models do not account the effect of non-hydraulic factors (e.g. temperature, riparian conditions and food availability) on the survival and ultimately future population trajectory (Bondi et al., 2013).

Another weakness occurring when using ecological models for hydropower dam impact assessment is non-transparent variable selection. In the reviewed papers, models often use just a few hydraulic (velocity, flow) and hydromorphological (water depth, substrate) characteristics to define habitat suitability. For instance, Pert and Erman (1994) simply used water depth and average velocity in a 20 m reach of river to define habitat preference for rainbow trout due to the difficulty to collect information on different habitat variables. Although the outcome of the model was valid, the authors suggested to consider other factors related to hydropower dam operation such as time during the day (morning, afternoon) and long term changes in discharge. It is hard to explore the effects of hydropeaking because it is strongly site specific and also depending the highly dynamic interactions between hydrology, hydraulics and morphological changes that can be hardly simulated (Boavida et al., 2015). Li et al. (2011a) used the average annual discharge for model development, but they suggested that the variation of hydrologic frequency as well as the maximum and minimum amount of water release from dams due to hydro-peaking should be taken into account. In many cases, there will be an overlapping effect of general river regulation impacts and hydropeaking impacts, being difficult to isolate the stressors (Boavida et al., 2015). Fish habitat simulations show that hydropeaking impact is strongly dependent on river morphology (Boavida et al., 2015).

Environmental impacts of dams affect different life stages and different groups, which requires a different level of detail in model development. Nevertheless, most model approaches base their assessment on physical-chemical water quality or the habitat suitability of adult fish (e.g. Ruetz and Jennings, 2000 ; Freeman et al., 2001 ; Cerny et al., 2003 ; Grand et al., 2006 ; Hatten et al., 2009 ; Garcia et al., 2011 ; Li et al., 2011b ; Cioffi and Gallerano, 2012 ; Chen et al., 2013 ; Wang et al., 2013), including often purely the most common and economic valuable species (e.g. trout, salmon) or endangered species. One publication considered the impact of flow regulation on algae (Wu et al., 2010), one paper focused on the effect on habitat suitability of amphibians (Yarnell et al., 2012) and just three papers considered macroinvertebrates (Gore and Hamilton, 1996; Li et al., 2009; Wang et al., 2013) in their assessment.

Another weakness is the lack of clear statistical criteria to assess the model fit. In addition, baseline data quantifying the ecological status prior to the building of the dam are often not available, which can result in lack of data for model development and model validation. Moreover, there is no standard sampling protocol available to allow a standardized assessment of the potential impact of a hydropower dam which is a crucial for international comparability.

c. Opportunities

An important opportunity related to the improvement of model prediction capacity is the increasing environmental data availability and data quality. The development of advanced environmental monitoring technologies could result in datasets with a large number of variables with high quality in order to deal with data scarcity and variability when developing models (Li et al., 2011a). Many European countries have established environmental monitoring networks to report on their river water quality. Apart from increased data availability, there has been substantial progress made in ecological modeling with regard to low quality or quantity data. When the amount of data is limited, datasets could be split into a training and validation set (Goethals et al., 2007) in order to test the robustness of the models. Moreover, independent experts could check model performance or the prediction of the model could be evaluated if the model output lies within the reliable ecological limits (Everaert et al., 2013). Furthermore, the combination of lab results and ecological models (Boets et al., 2010) and integration of data-driven models (Hoang et al., 2013) can be used to support decision-making in water management. When models are used for scenarios' analysis for which no data exists, a possible way to validate predictions may be through creating different models of the same system and then comparing predictions between models (Parrott, 2011), typically applied for environmental impacts assessment before hydropower dam construction.

To solve the problem of the single impact approach, an integrated model can be applied. For instance, an integrated model which considers physical-chemical, hydraulic and hydro-morphological characteristics could be used to assess the multiple effects related to flow variation on the river ecosystem (Holguin-Gonzalez et al., 2013; Holguin-Gonzalez et al., 2014). Models can be used qualitatively and quantitatively to consider climate change impacts on hydropower systems for hydropower relicensing (Rheinheimer et al., 2013). Given their numerous strengths and opportunities, using models for hydropower dam impact assessment deserves further exploration, especially, for the future the potential to use integrated models and to improve data collection to reduce the disadvantages of existing models or modeling techniques in hydropower dam impact assessment. Based on the above-mentioned opportunities in combination with technical advances in modeling it is believed that ecological models can be widely used to support hydropower dam impact assessment.

Considering these complicated processes present in ecosystems in the model does not mean that we need to make an overly complex model that takes a long computational time and that is not transparent for decision makers. A possible solution is the development of an integrated model instead of a single disciplinary model approach with complex input variables. Using an integrated model allows getting insight into the processes that occur on the river besides the direct impacts observed related to dam construction or wastewater discharges (Holguin-Gonzalez et al., 2014).

d. Threats

The management strategies solely developed based on models results could potentially pose a serious threat, because the output derived from a model can sometimes be inaccurate or deviating from reality. Li et al. (2009) pointed out that the physiology of organisms varies over different life stages, and caution is needed when applying year-round data. Therefore, using one model for different life stages and for the whole year conditions may lead to incorrect prediction results. Besides hydraulic characteristics, ecological components (e.g. riparian vegetation, present of predators, life stage of organisms), morphological characteristics (e.g. spectrum of large river systems of the river) (Enders et al., 2009; Yarnell et al., 2012) and physical conditions (e.g temperature, turbidity, substrate, seasonal variation…) (Ruetz and Jennings, 2000; Li et al., 2011b) are important factors, which should be taken into account. The study of complex systems thus requires a multi-scale approach, in which it is needed to consider the interactions occurring across many scales of space, time and organization (Parrott, 2011). Modelers should consider all relevant disturbances and predict various types of impacts before presenting the model output of ecological impact assessment of dams. Nevertheless, stakeholders involved in decision-making processes (e.g. policy makers, water managers, modelers…) need to be aware of the uncertainties of the model outputs and the risks/impacts these entail for considered actions.

Most of the models to study hydropower dam impacts were developed for particular organisms or a specific location. The habitat suitability criteria used, might be suitable for a specific case, but it may not be useful for other locations; therefore, it is needed to validate the habitat suitability model on independent data originating from different geographic locations (Yarnell et al., 2012). For instance, models developed to assess the impact of small hydropower dams in mountainous streams might not be applicable to assess the impact on large lowland rivers (Li et al., 2009). As a result, the choices of input variables and derived ecological indicators need critical review and validation by local experts before being reliable for model development. An assessment of hydropower dams based on models must be made with great care, because models cannot always separate the dam impact from other anthropogenic influences (de Merona et al., 2005).

2.4 Conclusions

Hydropower dams affect the hydromorphological, physical-chemical and biological conditions in rivers. The models applied to hydropower dam impact assessment have been used to assess an expanding range of impacts at various spatial and temporal scales. Existing ecological models provide a basis to assess the impact of changing hydrological regimes and water quality on the habitat suitability of fish, macroinvertebrates and algae. Empirical data and knowledge have been used as input variables for hydrodynamic models, water quality models, food web models, habitat suitability models and integrated models to explore the change of water quality and habitat suitability at various scales. Although a few models combine two or more model types, no integrated model has been proposed so far. Given their numerous strengths and opportunities, using models for hydropower dam impact assessment deserves further exploration and optimisation, especially, for the future the potential to use integrated models and to improve data collection to reduce the disadvantages of existing models or modeling techniques in hydropower dam impact assessment.

Chapter 3: Materials and methods

Chapter 3: Materials and methods

Abstract

This chapter provides an overview of the two river basins in which the data were collected. The Guayas River basin is located in the central-western part of Ecuador and is the largest watershed in the Guayas province. The Portoviejo River basin is located along the coast in the western part of Ecuador. Both river basins are affected by the installation of hydropower dams, but also water pollution is in general present in different locations. Also the monitoring methods and the assessment methods are described, in particular related to the hydromorphological inventarisation, the physical-chemical measurements as well as the invertebrates and water hyacinth inventarisations. The last part deals with the assessment and modeling methods. Both Generalized Linear Models (GLMs) and Threshold Indicator Taxa ANalysis (TITAN) were applied. The latter was in particular used to detect changes in taxa distributions along an environmental gradient over space or time.

3.1 Study areas

3.1.1 The Guayas River basin

The Guayas River basin, with a total area of 34,000 km^2, is located in the central-western part of Ecuador and is the largest watershed in the Guayas province (Arias-Hidalgo, 2012). The Eastern catchment boundary of the basin is the Andes. The Guayas basin consists of two large tributaries being the Daule River and the Babahoyo River (Arias-Hidalgo, 2012). The Guayas River is the mouth of the Guayas river basin, which is formed by the confluence of two tributaries (Fig. 3.4). The mouth of the basin is situated in near Guayaquil, a city which is located along the Ecuadorian coast, where the river discharges into the Pacific Ocean at the Gulf of Guayaquil.

The Daule-Peripa hydro-electrical project is located in the upper catchment of the Guayas river basin. The Daule-Peripa reservoir was constructed in 1987 and receives water from the Daule and Peripa Rivers (Gelati et al., 2011) and is used for hydropower generation, irrigation, flood protection and drinking water (Arriaga, 1989). The reservoir has a water storage capacity of 6,000 million m^3, a maximum surface area of approximately 30,000 ha and the water depth fluctuates between 70 and 85 m due to the operation of the dam (CELEC, 2013).

The seasons are well defined in Ecuador, in the particular basins: the rainfall is concentrated in the wet season (December-April) and the dry season months are May-November (Camposano, 2004). The mean temperature and annual rainfall of the region vary between 22 ºC and 27 ºC and 300-4000 mm, respectively (Madonado, 2011).

Figure 3.1 Picture taken at the Daule-Peripa hydroelectric reseroir

3.1.2 The Portoviejo River basin

Portoviejo is the capital of the Province of Manabí (Ecuador) and is situated 30 km from the Pacific coast. The Portoviejo River basin, with a total area of 2,231 km^2 and 132 km long (Pérez, 2003b), is located along the coast in the western part of Ecuador (U.S. Army Corps of Engineers, 1998). It discharges water into the Pacific Ocean at La Boca. The river provides water to 700,000 inhabitants for domestic use, agriculture, recreation and other purposes (Pérez, 2003b). The Portoviejo River basin is one of the most productive farming regions in Ecuador, with production of bananas, mangoes and other tropical fruits, tomatoes, onions, peppers, coffee, and especially cattle and fish (http://www.gutenberg.us, 2016). The Poza Honda dam is located 30 kilometer upstream from the city of Portoviejo and started operating in 1971 (Fig. 3.2). The Poza Honda reservoir has a water storage capacity of 100 million m^3 (U.S. Army Corps of Engineers, 1998), a maximum surface area of approximately 607,5 ha and a maximum depth of 37.3 m. The Poza Honda reservoir faces eutrophication problems due to intensive agriculture and livestock (Perez, 2004) in the surrounding area.

Figure 3.2 Picture taken at the tailrace of the Poza Honda hydropower dam.

Portoviejo has a low latitude and a semi-arid hot climate. The seasons are well defined; the rainfall is concentrated in the period December-May, in which 90% of the annual rainfall occurs and the dry season months are June-November. The mean temperature and monthly rainfall of the region vary between 24ºC and 29ºC and 2 to 115 mm, respectively (http://www.portoviejo.climatemps.com). Land use in the basin consists mostly of arable land, plantations (onions, bananas, and other tropical fruits), urban and semi-urban areas.

3.2 Data collection

During designing the sampling campaign, several aspects were considered, which included (1) appropriate number of sampling sizes, (2) including environmental information in the study areas and (3) applying random sampling. Seasonality was considered when designing the sampling campaign; we assumed the worst-case conditions in terms of water quality (e.g. conductivity) during the dry season (low dilution due to rain), possibly indicating severe water quality problems. In that way, a single observation of maroinvertebrates is suitable for the ecological water quality assessment purpose. Environmental variables that are expected to have a strong effect on macroinvertebrates were selected. It was expected that the ecological water quality would decrease from source to mouth due to increasing human pressure.

Therefore, samples should be taken at disturbance sites and the less human impacts sites (e.g in the mountainous area/upstream reaches).

Hydropower dam was only one of the major factors that affect the river water quality. In both Guayas river basin and Portoviejo river basin, the main identified sources of disturbance are damming, residential areas, wastewater discharges and agricultural activities (Fig. 3.3). Therefore, the standard assessment approach was considered and the following selection criteria were applied:

- including different impacts and various levels of water quality;
- including sites from upstream of the dam, reservoir and downstream of the dam;
- distributing sites over the entire basin;
- being relatively easily accessible.

Figure 3.3 Major disturbances observed in the Ecuadorian rivers: (A) human disturbances, (B) urbanization and land use change, (C) agriculture activities, (D) domestic wastewater discharge.

3.2.1 Data collection in the Guayas River basin

In total, 120 sampling sites were selected in the Guayas River basin, of which 32 were located in the Daule-Peripa reservoir (detailed descriptions in Fig. 3.5 and Fig. 3.6) and the remaining 88 locations were located along the rivers within the Guayas River basin, including upstream and downstream locations (Fig. 3.4). The sites were selected along the Daule River, the Babahoyo River and at the main tributaries with an expected gradient of disturbance from upstream (mountainous areas with fewer human impacts) to downstream (lowland with more human impacts).

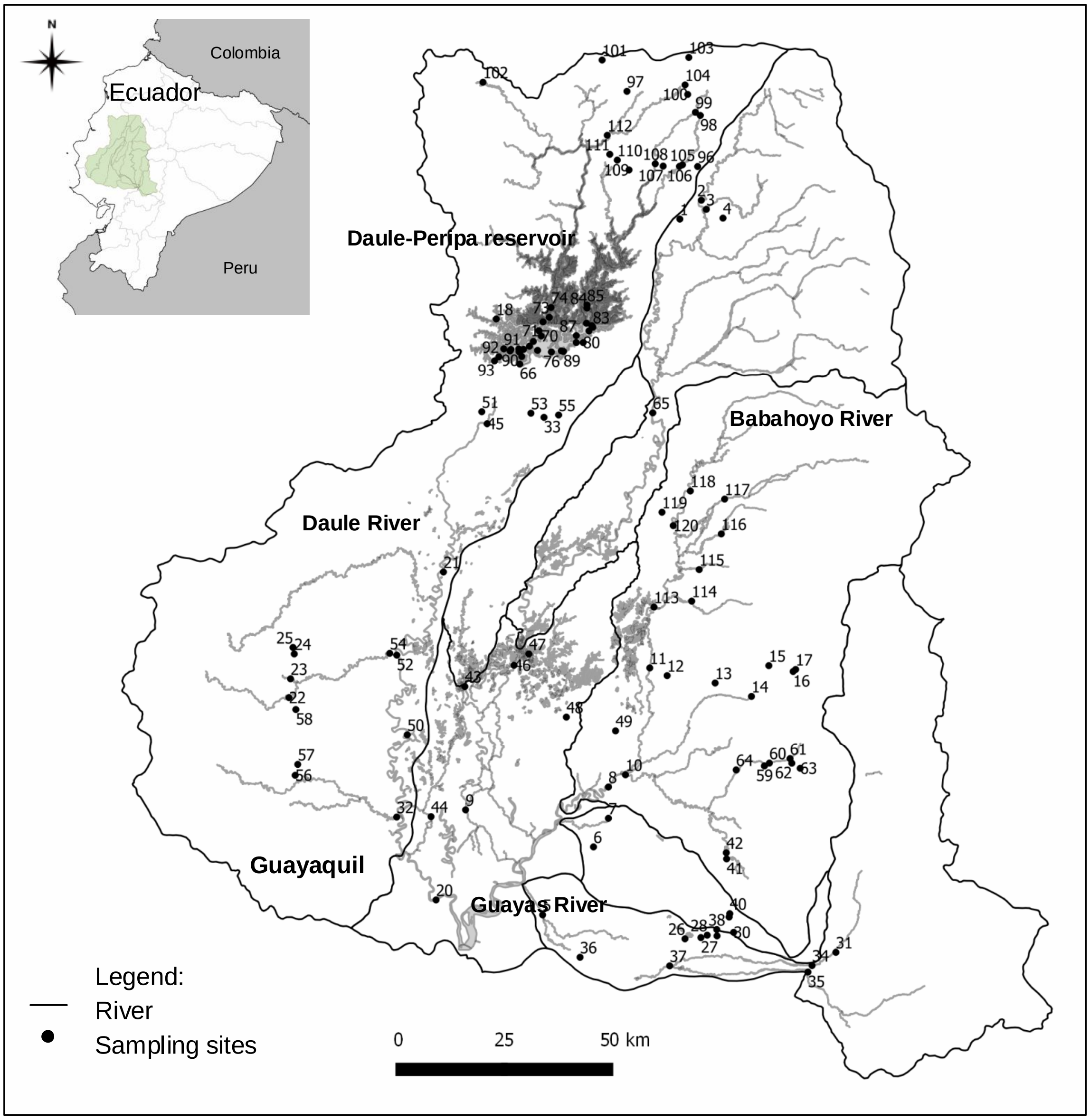

Figure 3.4 Map of the study area in the Guayas river basin (numbers indicate of sampling sites)

The water samples were collected in the dry season of 2013. Temperature (°C), pH, Dissolved Oxygen (DO) (mg/L), Chlorophyll a (μg/L), Chloride (mg/L), Turbidity (FTU), Conductivity (μS/cm) and Total Dissolved Solids (TDS) (mg/L) were measured at the surface water layer with a multiprobe (model YSI 6600 V2 and YSI

6600 V1, YSI manufacturer). Moreover, water samples from each sampling location were collected and stored in plastic bottles (1l), kept cool and dark immediately after collection and transported to the laboratory for further analysis. At the laboratory, the water samples were kept in a refrigerator (cool and dark) for nutrient analysis. In the laboratory, the Hach Lange DR 3900 photometric method was used to determine chemical oxygen demand (COD) (mg/L), total phosphorus (TP) (mg/L), ammonium (NH_4^+-N)(mg/L), nitrate (NO_3^- -N)(mg/L), nitrite (NO_2^- -N)(mg/L) and total nitrogen (TN) (mg/L). However, the values of all variables analysed at the laboratory except for ammonium were below the detection limit of the cuvette test on Hach Lange DR 3900. Consequently, those variables were not included in the analysis. The lowest detection limit of COD was 5 mg/L, TN was 1 mg/L, TP was 0.5 mg/L, NO_3^- -N was 0.23 mg/L, NO_2^- -N was 0.015 mg/L and NH_4^+-N was 0.015 mg/L.

The elevation of sampling sites was measured using GPS (Global Positioning System) equipment (Garmin GPS). Surface water velocity was measured using the techniques for estimating stream velocity described in United States Environmental Protection Agency (1997).

In the Daule-Peripa reservoir, 32 samples were taken, of which eleven samples located at non-vegetated sites and 21 sites were characterised by vegetation cover (the presence of waterhyacinth) (Fig. 3.3 and Fig. 3.4). At 21 vegetated sites, water hyacinth was the only floating macrophyte occurring. Only at one vegetated site a native emergent plant species (*Sagittaria* sp) was recorded at very low abundance. The percentage of water hyacinth cover was visually estimated from the bank along a transect of 100m; the location where water samples were taken was considered as the centre of the transect. The vegetation cover classes were divided according to the Braun-Blanquet cover/abundance scale: 0 = non-vegetated/absent, 1 = 1-5% (rare), 2 = 5-25% (occasional), 3 = 25-50% (frequent), 4 = 50-75% (common) and 5 = 75-100% (abundant). Classes rather than exact values for cover of water hyacinth were used as this yields more reliable measures (Ellenberg and Mueller-Dombois, 1974). Based on the vegetation cover, sampling sites were classified as follows: class 0: 11 sites, class 1: 5 sites, class 2: 0 sites, class 3: 2 sites, class 4: 3 sites and class 5: 11 sites.

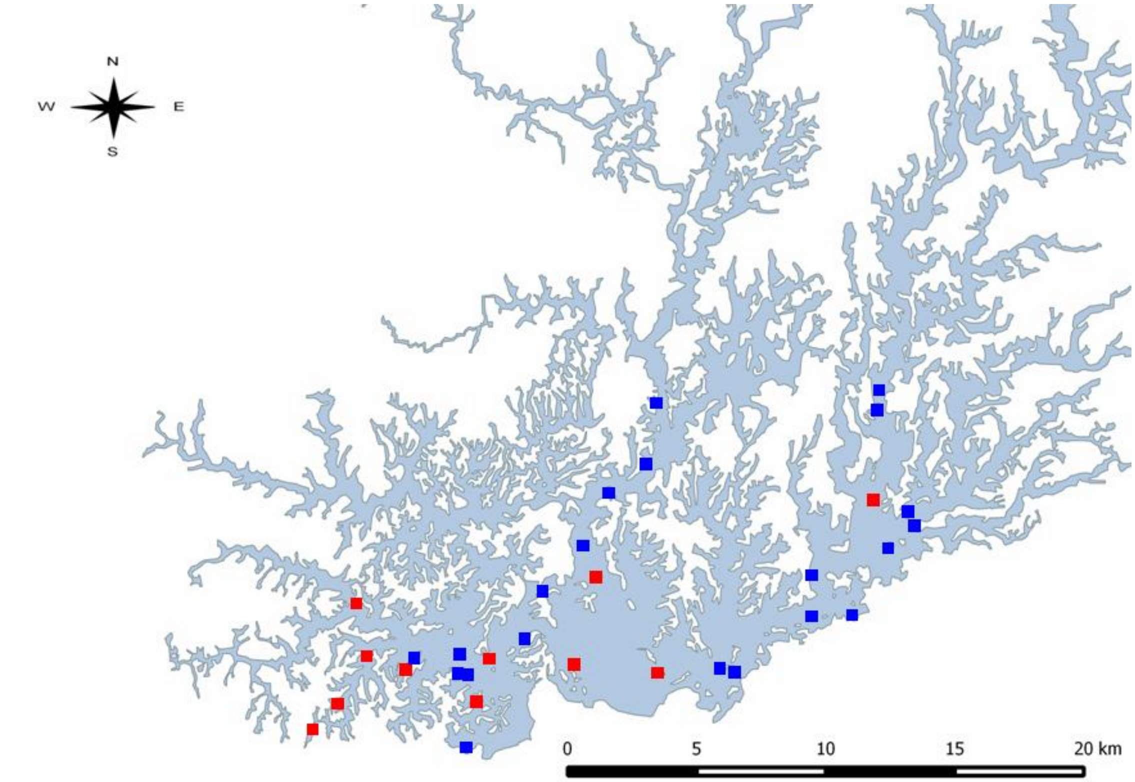

Figure 3.5 Map of study area with indication of the sampling sites and the presence (blue) or absence (red) of water hyacinth in the Daule-Peripa reservoir, Ecuador.

Figure 3.6 Picture taken at the Daule-Peripa dam of a site completely covered by water hyacinth.

3.2.2 Data collection in the Portoviejo River basin

In the Portoviejo River basin, 31 sampling sites were taken in the dry season of 2015 (Fig 3.7). Eighteen sites were taken starting from 10 km upstream of Portoviejo city to the mouth of the river. These sites were considered as more impacted sites. Other13 sites located from 20 km upstream of Portoviejo city to the source of the river. These sites were chosen as less impacted sites and serve as reference locations.

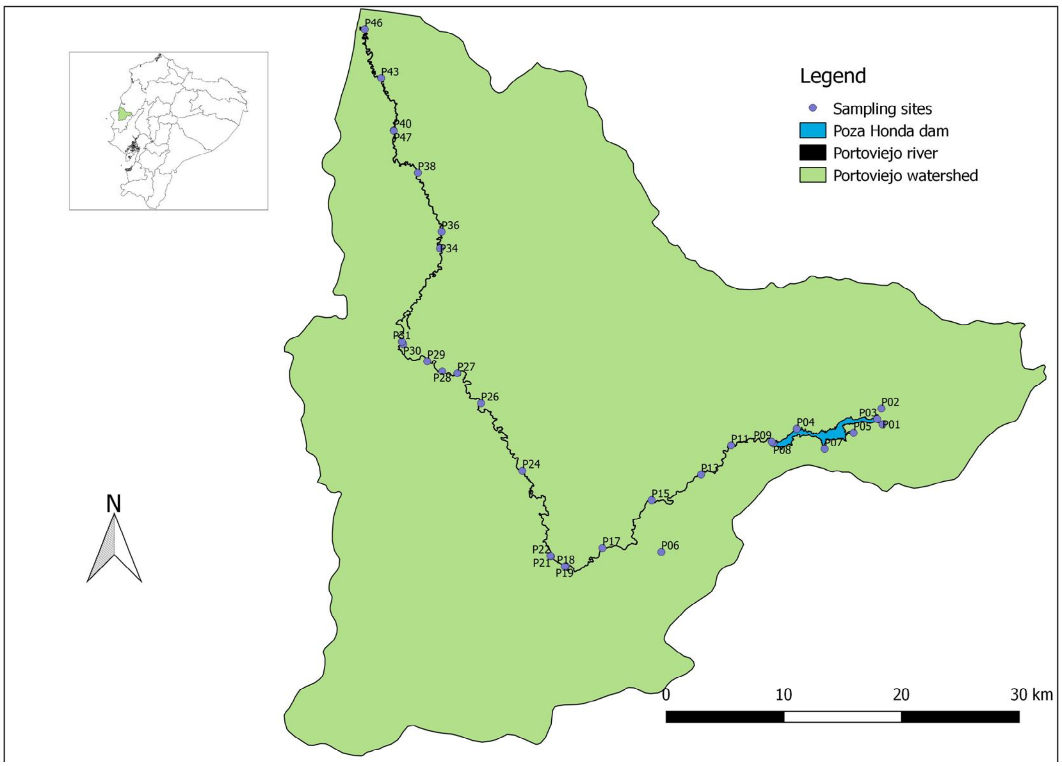

Figure 3.7 Map of the study area of the Portoviejo River with indication of the sampling sites

Values of temperature (°C), pH, dissolved oxygen (DO) (mg/l), chlorophyll a (µg/l), turbidity (FTU) and electrical conductivity (EC) (µS/cm) were measured at the water surface with a multiprobe (model YSI 6600 V2, YSI manufacturer). Moreover, water samples from each sampling location were collected and stored in plastic bottles at each sampling site (one liter), kept cool and in the dark immediately after collection and transported to the laboratory for further analysis. At the laboratory, the water samples were kept in a refrigerator (cool and dark) for nutrient analysis. The

Hach-Lange DR 3900 spectrophotometer kits were used to determine biological oxygen demand (BOD_5) (mg/l), total phosphorus (TP) (mg/L), orthophosphate (oPO_4^{3-}) (mg/L), ammonium (mg/L), nitrate (NO_3^-) (mg/L), nitrite (NO_2^-) (mg/L), total nitrogen (TN) (mg/L) and total organic carbon (TOC) (mg/L).

The elevation of sampling sites was measured using GPS (Global Positioning System) equipment (Garmin GPS). Stream velocity was measured with a handheld flow meter (HFA, Höntzsch, Waiblingen, Germany). The surrounding land use was determined and divided into five classes (shrubs/grasses, orchard, residential, arable land and forest). The type of dominant substrate was visually assessed at each site and divided into five classes (silt or clay, sand, gravel, cobble and boulder). The sludge layer was classified into absent, <5cm, 5-20 cm and > 20cm. Six classes of the pool-riffle pattern were distinguished (absent due to structural changes, absent, poorly developed, moderately developed, well developed and pristine). The hydro-morphological characteristics of the sampling sites were determined based on field inspection and completed per sampling location by a standard field protocol. The field protocol was modified from the Australian River Assessment System (AUSRIVAS) physical assessment protocol (AUSRIVAS, 1994) and the United Kingdom and Isle of Man River Habitat Quality (Raven et al., 1998) (Appendix 3.1).

3.2.3 Macroinvertebrate community monitoring and analysis

Macroinvertebrates were collected with a standard hand net consisting of a metal frame holding a conical net (mesh-size 300 µm) at the same sites where water quality was measured. Macroinvertebrates were collected during five minutes active sampling, including all different microhabitats present at the sampling site (Gabriels et al., 2010). At sampling sites in the Daule-Peripa reservoir, at the non-vegetated sites samples were collected by vigorously sweeping the net along the reservoir margins and by disturbing the bank substratum at wading depth. For those sites that contained macrophytes, macroinvertebrates were obtained by submerging the hand net under the roots of the macrophytes andby disturbing the stalks and leaves. Afterwards, the net was quickly and carefully lifted out of the water to prevent the escape of mobileorganisms. Macroinvertebrates attached to the roots of water hyacinth were manually collected. Similarly, macroinvertebrates were collected from stones and other substrates at both vegetated and non-vegetated sites (Gabriels et al., 2010).

Samples collected were sieved (500 μm mesh size) in the laboratory and sorted in white trays. Macroinvertebrates from each location were placed in separate small plastic vials containing 80% ethanol for preservation. After sorting, organisms were identified and counted under a stereomicroscope.

Macroinvertebrates were identified to family level using the identification keys developed by Domínguez and Fernández (2009) for three reasons. Firstly, previous research has shown that using biotic indices based on family level provides sufficient information to assess the biological water quality (Dominguez-Granda et al., 2011b; Mereta et al., 2013; Everaert et al., 2014). Secondly, family-level identifications have been found as useful as species-level for bioassessment (Marchant et al., 2006). Thirdly, because of practical implications we could only identify up to family level as there are no detailed keys available to lower taxonomic levels.

3.3 Ecological modeling and assessment methods

3.3.1 Generalized linear models (GLMs)

Generalized linear models (GLMs) was formally introduced by Nelder and Wedderburn (1972). GLMs are mathematical extensions of linear models that do not force data into unnatural scales, as thus GLM allows for non-linearity and non-constant variance structures in the data (Hastie and Tibshirani, 1990). GLMs are based on an assumed relationship (link function) between the mean of the response variable and the linear combination of the explanatory variables. Data may be assumed to be from several families of probability distributions, including the normal, binomial, Poisson, negative binomial, or gamma distribution, many of which better fit the non-normal error structures of most ecological data (Guisan et al., 2002). Thus, GLMs are more flexible and better suited for analyzing ecological relationships, which can be poorly represented by classical Gaussian distributions (Guisan et al., 2002).

GLMs have been broadly applied in ecology. Some general uses of GLMs in studies of species distributions are discussed in Guisan et al. (2002). The overview of theory for the applications of GLMs in fisheries research can be found in (Venables and Dichmont, 2004). Some specific examples includes the use of GLMs to predict the potential distribution of plant species (Guisan et al., 1998), of fish (Fukushima et al., 2007) and of bird habitat suitability (Brotons et al., 2004). In this PhD study, GLM was applied to investigate which variables determined the occurrence of water hyacinth in the Daule-Peripa reservoir.

3.3.2 Threshold Indicator Taxa ANalysis (TITAN)

Baker and King (2010) introduced a technique called Threshold Indicator Taxa ANalysis (TITAN) to detect changes in taxa distributions along an environmental gradient over space or time. TITAN is a non-parametric technique, which uses indicator species scores to integrate occurrence, abundance and directionality of taxa responses. TITAN was used to analyze the ecological data (community composition and abundance at each sampling location) along each continuous predictor variable gradient. TITAN partitioned sample units into two groups at the value of a predictor variable that maximizes the association of each taxon with either the negative or positive side of the partition. The association is measured by indicator values (IndVal), which are calculated for all species for all possible change points along the environmental gradient, with permutation tests to assess the uncertainty in these scores. TITAN distinguishes negative (z-) (decreasing frequencies and abundance at the changing point) and positive (z+) (increasing frequencies and abundance at the changing point) taxa responses. The medians of multiple taxa changes were defined as entire community thresholds. Two important diagnostic indices measuring the quality of the indicator response for any taxon which are obtained from bootstrap resampling are purity and reliability. *'Purity'* is the proportion of change-point response directions (positive or negative) among bootstrap replicates that agree with the observed response. The 'pure indicators' are consistently assigned the same response direction, regardless of abundance and frequency distributions generated by resampling the original data. '*Reliability*' is estimated by the proportion of bootstrap change points whose IndVal scores consistently result in *p*-values below one or more user-determined probability levels. Permuted IndVal scores are standardized as z scores and summed for positive [sum(z+)] and negative value [sum(z−)] responses for each possible change point. TITAN was performed in the package TITAN 2 (Baker et al., 2015) in R software (version R.3.2.3). Only taxa occurring in at least five sites were included in TITAN (Baker et al., 2015). Taxa names which have more than eight characters were coded as eight character abbreviations of scientific names. Each environmental variable was used as a predictor in separate models. Abundance data was not transformed because transformation is unnecessary in TITAN 2. In this study, 1000 repetitions (Bootstrapping) was implemented to estimate uncertainties around change point locations and response direction (positive or negative) of threshold indicator taxa. 1000 permutations were used to determine species specific z scores, as this calculation is based on a small dataset, thus a higher number of permutations are recommended for more precise z scores (Baker et al., 2015). Details of the TITAN method can be found in Baker and King (2010), King and Baker (2014) and Baker et al. (2015).

There is an increasing interest in the application of ecological thresholds for natural resources management (King and Baker, 2010). In previous studies, TITAN has been used to detect thresholds for benthic invertebrates along a gradient of chloride concentrations (Wallace and Biastoch, 2016), percent glacier cover and percent melt water in the catchment (Khamis et al., 2011), percent vegetation loss (Rodrigues et al., 2016) and percent impervious cover (King et al., 2016). Other studies used TITAN to identify thresholds for benthic invertebrates and diatom communities along a gradient of salinity (Schroder et al., 2015). Additionally, some studies used TITAN to determine the response of the macroinvertebrate communities to total phosphorus (Baker and King, 2010) or the response of the phytoplankton communities to nutrient gradients (e.g. total phosphorus (TP) (Smuckera et al., 2013; Cao et al., 2016), total nitrogen (TN) and TN:TP ratios (Cao et al., 2016). Berger et al. (2016) used TITAN to look for the relationships between benthic invertebrates and various chemical variables related to pesticides, wastewater and fossil fuel-associated chemicals. Other TITAN applications include response analyses of macroinvertebrate, fish, bird, diatom and wetland vegetation communities to changes of the landscape (Kovalenko et al., 2014). Species-specific thresholds provide an indication of whether and when species are likely to be affected by changing environmental conditions (Schroder et al., 2015). TITAN reveals the ecological community threshold at which the abundance or frequency of taxa will quickly increase or decrease along an environmental gradient (King and Baker, 2014). Therefore, from a conservation point of view, it is important to understand which taxa are affected by environmental stressors and how threshold values for river biota can be identified. Information about species and community thresholds is useful to further delineate the conservation value of a sensitive species or to predict the changes of community composition (Schroder et al., 2015). TITAN is able to inform managers about critical levels of anthropogenic changes that are related to rapid changes in ecological communities (King and Baker, 2014). The TITAN results have valuable applications for detecting reference condition boundaries and selecting areas at the greatest risk of significant change (Kovalenko et al., 2014). This information could be used for aquatic conservation, biological invasions, ecosystem restoration and natural resource management (King and Baker, 2010).

In this study, TITAN was used to detect the responses of macroinvertebrate community to the changes of environmental gradients in both Guays River basin and Portoviejo River basin.

3.4 Assessment approaches based on macroinvertebrates

The use of macroinvertebrates as bio-indicators for freshwater quality has a long history. There are several assessment methods based on macroinvertebrates have been developed worldwide for stream assessment. Overviews of the existing assessment approaches can be found in De Pauw et al. (2006). In this part, a brief overview of the three approaches used in this study is presented.

3.4.1 Diversity approach

The diversity approach uses three components of community structure, namely, richness (number of species present), evenness (uniformity in the distribution of individuals among the species) and abundance (total number of organisms present) to describe the response of a community to the quality of its environment (Metcalfe, 1989). The assumption is that undisturbed environments are characterized by a high diversity or richness, an even distribution of individuals among the species, and moderate to high counts of individuals. In contrast, the disturbance of the water ecosystem leads to a reduction in diversity. The advantages and limitations of diversity approach have been reviewed by Metcalfe (1989): The advantages include their easiness to use and calculate, applicability to all kinds of watercourse with no geographical limitations. Moreover, diversity indices are strictly quantitative, dimensionless and rely on statistic analysis and are best suited for comparative purposes (Metcalfe, 1989). In addition, diversity indices are independent of sample size and are applied equally to measures of biomass (Metcalfe, 1989). However, diversity index values are unable to indicate if the community consists of pollution-tolerant or pollution-intolerant species. Furthermore, diversity index values vary greatly since they depend on the sampling method and the nature of the study site (Metcalfe, 1989). In this PhD study, several diversity indices have been calculated, include, Shannon-Wiener Diversity Index (Shannon and Wiener, 1949), Evenness index (Menhinick, 1964), Simpson's diversity index (Simpson, 1949) and Margalef index (Margalef, 1968).

3.4.2 Biotic approach

The biotic approach, which is combining a quantatitive measure of diversity with the qualitative information on the ecological sensitivities of individual species or higher taxa or groups into a single index or score. The principle of biotic approach is that macroinvertebrate groups disappear as pollution increases and that the number of taxonomic groups is reduced as pollution increases (Mackenthun, 1969). The advantages of biotic approach are that only qualitative sampling is required and that identification is mostly at family or genus level and that there is no need to count abundances per taxon (De Pauw et al., 2006). However, the remaining constraints are determining representative reference communities to which the investigated stations can be compared and optimising biological assessment through regional adaptations (De Pauw et al., 2006).

The Biological Monitoring Working Party (BMWP) (Armitage et al., 1983) is the first biotic index developed for the assessment of running water, which is the water quality index used in the United Kingdom. Based on a biotic approach and predominantly adaptations of the English BMWP index, several bioassessment methods have been developed for tropical regions. Examples includes the BMWP was adapted for Colombia (Pérez, 2003a), BMWP adapted for Costa Rica (Astorga et al., 1997) and BMWP adapted for Thailand (Mustow, 2002). Colombia is the country which is closest to Ecuador and having similar environmental conditions and the fauna to Ecuador (Dominguez-Granda et al., 2011b). As Ecuador does not have its own water quality index but Colombia has biological indices. Therefore, the BMWP-Colombia was considered as appropriate index for Ecuador.

In this PhD study, the assessment of the water quality was based on the biotic macroinvertebrate index BMWP-Colombia (Biological Monitoring Working Party-Colombia). The BMWP-Colombia index was calculated according to the modified method proposed by Zuniga and Cardona (2009). The BMWP-Colombia was calculated per site based on a summation of all tolerance scores of the macroinvertebrate taxa present. Each macroinvertebrate taxon received a score that reflects its susceptibility to pollution, where pollution-intolerant taxa receive high scores, whereas pollution-tolerant taxa were given low scores (Zuniga and Cardona, 2009). The total score for each site indicated the water quality, with categories ranging from very bad (0–15), bad (16–35), poor (36–60), moderate (61–100) to good (>100).

3.4.3 Multivariate approach

Several multivariate techniques have been used in water quality assessment using macroinvertebrates, of those classification, ordination, and discriminant analysis are some of the most widely used techniques (Norris and Georges, 1993). The basis for the multivariate approach is the similarity index (Sandin et al., 2001). The most commonly used similarity index is the Jaccard index, which expresses the percentage of species shared between two sites (Sandin et al., 2001). Other examples are the Bray–Curtis dissimilarity index (Bray and Curtis, 1957), and the Euclidean or ecological distance (Williams, 1971). The similarity indices can be used to calculate the distance of the biological assemblage at each sampled site from the median of all reference communities (Sandin et al., 2001). Multivariate methods allow the correct selection of metrics for multimetric systems, avoiding the use of redundant variables. In this sduty, the Non-metric Multidimensional Scaling (NMDS), Detrended Correspondence Analysis (DCA) and Canonical Correspondence Analysis (CCA) ANOSIM (ANalysis Of SIMilarities) (based on the Bray-Curtis similarity), A SIMPER (SIMilarity PERcentages) were used.

Chapter 4: Habitat suitability of the invasive water hyacinth and its relation to water quality and macroinvertebrate diversity in a tropical reservoir

Adapted from:

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Chapter 4: Habitat suitability of the invasive water hyacinth and its relation to water quality and macroinvertebrate diversity in a tropical reservoir

Abstract

In this chapter, we assessed the relationship between the occurrence of the invasive water hyacinth (*Eichhornia crassipes*) and water quality properties as well as macroinvertebrate diversity in a tropical reservoir, situated in western Ecuador. Macroinvertebrates and physico-chemical water quality variables were sampled at 32 locations (during the dry season of 2013) in both sites covered and non-covered by water hyacinth in the Daule-Peripa reservoir. The results indicated that, in terms of water quality, only turbidity was significantly different between sampling sites with and without water hyacinth (Mann-Whitney U-test, $p<0.01$). Higher turbidity was observed at non-vegetated sites. The habitat suitability model showed that water hyacinth was present at sites with a low turbidity but this model is merely generated based on 'correlations'. The cause-effect relationships between the presences of water hyacinth and other environmental variables was not investigated. The Biological Monitoring Working Party-Colombia score and the Margalef diversity index were significantly higher (Mann-Whitney U-test, $p<0.01$) at sampling sites where water hyacinth was present compared to water hyacinth absent sites. However, there were no significant differences in the Shannon–Wiener index, Evenness index and Simpson index between the sampling sites with and without water hyacinth. Our results suggest that water hyacinth cover was an important variable affecting the diversity of macroinvertebrates in the Daule-Peripa reservoir, with intermediate levels of water hyacinth cover having a positive effect on the diversity of macroinvertebrates. Information on the habitat suitability of water hyacinth and its effect on the physico-chemical water quality and the macroinvertebrate community are essential to develop conservation and management programs for large tropical reservoirs such as the Daule-Peripa reservoir and the Guayas river basin, where water resources are being at high risk due to expansion of agricultural and industrial development activities.

4.1 Introduction

Alien invasive plant species, mainly of terrestrial habitats, have recently received much attention due to their direct or indirect impact on ecosystem structure and functioning (Vilà et al., 2011). The impact of plant invasions in aquatic ecosystems has received less attention. Several authors reported that invasive plant species can cause serious ecological and economic impacts such as loss of native species diversity, hybridization with native species, changes in ecosystem processes and functioning, and an increase of pests and diseases (Rodriguez, 2006; Stiers et al., 2011; Walsh et al., 2012). Alien invasive plant species may alter the available structure in aquatic habitats by creating a shift to a homogeneous habitat, there by negatively affecting biological communities (Theel et al., 2008; Schultz and Dibble, 2012). Indeed, simplification of the macrophyte growth form seems to negatively affect the abundance of biotic communities such as macroinvertebrates (Walker et al., 2013). Schultz et al. (2012) found that mechanisms underlying the impact of aquatic invasive plants are not very different from native aquatic plants species. They identified three invasive traits largely responsible for negative effects on biological communities: increased growth rate, allelopathic chemical production and phenotypic plasticity. Despite the negative impact that is often observed, positive relationships between invasion by alien macrophytes and macroinvertebrate diversity have been observed as well (e.g. Brendonck et al., 2003; Villamagna, 2009).

Water hyacinth *Eichhornia crassipes* (Mart.) Solms is one of the most widely spread invasive aquatic macrophytes in the world (Villamagna and Murphy, 2010). The species has its origin in the Amazon basin and in the vast wetlands of the Pantanal in western Brazil (Parolin et al., 2012). Water hyacinth is a free floating aquatic plant, that can spread fast and has a strong growth (Malik, 2007). According to Malik (2007), water hyacinth can double its size (area covered) in five days and a mat of medium sized plants may contain two million plants per hectare that weigh 270 to 400 tons. Under favorable conditions the biomass of water hyacinth can be doubled within only 12 days (Parolin et al., 2012). Nowadays, water hyacinth is distributed worldwide (Parolin et al., 2012) and the International Union for Conservation of Nature listed the species as one of the 100 most harmful invasive species (Lowe S. et al., 2000).

Water hyacinth tolerates a wide range of nutrients, temperature and pH levels and grows in a wide variety of ecosystem types (Malik, 2007). Environmental factors such as temperature, pH, solar radiation, and salinity of the water can influence the growth and performance of water hyacinth (Gupta et al., 2012). The optimum conditions for growth of water hyacinth are a pH between 6 to 8 (Malik, 2007) anda temperature that ranges between 28 and 30°C (Gupta et al., 2012). A study by Wilson et al. (2005) showed that nutrient concentrations and temperature are two of the most important factors determining water hyacinth growth and reproduction. Salinity on the other hand can cause major constraints for water hyacinth growth, since water hyacinth can not survive at a salinity higher than 2‰ (Olivares and Colonnello, 2000).

Water hyacinth can have an effect on the physical and chemical composition of water. The introduction of water hyacinth can cause a change in water clarity, hydrological regime, dissolved oxygen concentration, nutrient concentrations and other pollutants in the water body (Villamagna and Murphy, 2010). Negative as well as positive effects have been recorded. Water hyacinth is known to reduce dissolved oxygen concentrations because the mats prevent the transfer of oxygen from the air to the water surface, while the plant does not release oxygen into the water (Meerhoff et al., 2003). Water hyacinth has the potential to stabilize pH and temperature and prevent stratification in lotic systems (Giraldo and Garzon, 2002) and in this way change ecosystem structure and functioning. Rommens et al. (2003) and Sooknah and Wilkie (2004) reported that water hyacinth has the capacity to absorb nutrients (e.g. nitrate, ammonium, phosphate) from the water column affecting phytoplankton and zooplankton abundance.

Water hyacinth plays a important role for phytoplankton, zooplankton and fish in freshwater ecosystems by providing habitat complexity, shelter and feeding grounds (Brendonck et al., 2003; Meerhoff et al., 2003; Villamagna and Murphy, 2010). According to de Marco et al. (2001) and Brendonck et al. (2003) the roots and the leaves of water hyacinth offer an important substratum and habitat for macroinvertebrate colonisation. Indeed, a positive relationship between macroinvertebrate diversity and the cover of water hyacinth has been found (Schramm et al., 1987; Kouame et al., 2011). Kouame et al. (2011) found a high diversity and density of macroinvertebrate assemblages associated with root masses of water hyacinth in Lake Taabo (Ivory Coast). They ascribed this to improvements in physico-chemical properties (e.g. conductivity, nutrients, temperature, turbidity), which positively influenced the macroinvertebrate community. Moreover, Bailey and Litterick (1993) also found a positive effect of water hyacinth on dissolved oxygen and a variety of potential food resources for aquatic invertebrates in water hyacinth root-mats resulting in higher abundances of macroinvertebrates. However, water hyacinth limits the development of phytoplankton via preventing light penetration and absorbing nutrients which reduces phytoplankton and leads to a decrease in zooplankton abundance (Villamagna and Murphy, 2010). In addition, water hyacinth causes significant ecological alterations in the invaded community by modifying the habitat, disrupting the food chain and nutrient cycling, and consequently changing invertebrate and fish assemblage structure and finally the entire food web (Brendonck et al., 2003; Toft et al., 2003). Hence, water hyacinth influences species richness, diversity and composition of invaded communities and causes huge impacts to ecosystems structure and functioning (Villamagna and Murphy, 2010).

Next to its ecological impact, water hyacinth has attracted global attention due to serious problems caused to power plants, navigation, irrigation andrecreation (Epstein, 1998; Lu et al., 2007; Malik, 2007). According to Malik (2007), many large hydropower plants have to spend considerable time and money in clearing water hyacinth in order to prevent it from entering in the turbines and causing damage and power interruptions. In China, water hyacinth is reported to clog lakes and rivers, impede water flows, obstruct navigation, and damage irrigation and hydroelectricity facilities, thus the total cost for water hyacinth control is estimated to be more than $12.35 million each year (Lu et al., 2007). Indeed, the invasion of water hyacinth causes huge ecological and economic consequences worldwide (Lu et al., 2007). However, their effects to human society vary according to the magnitude of invasion, the ecosystem services of the water body and the response of water hyacinth to management efforts (Villamagna and Murphy, 2010).

There have been several studies carried out on the effect of water hyacinth on water quality and aquatic diversity, the invasion mechanism and the utilization in temperate regions (Fan et al., 2013). However, until now the habitat suitability of water hyacinth and the relationship between water hyacinth and ecological communities largely remains understudied in tropical reservoirs. Therefore, the aim of this chapter was to determine the habitat suitability of water hyacinth and to assess the effect of water hyacinth cover on the chemical water quality and on the macroinvertebrate community in a tropical reservoir. This information is essential to develop conservation and management programs for tropical reservoirs and specifically the Guayas river basin, situated in western Ecuador, where water resources are being at high risk due to expansion of agricultural and industrial development activities.

4.2 Data analysis

In this chapter, we used the data from 32 sampling sites in the Daule-Peripa reservoir for analysis. The Shannon-Wiener Diversity Index (Shannon and Wiener, 1949), Evenness index (Menhinick, 1964), Simpson's diversity index (Simpson, 1949) and Margalef index (Margalef, 1968) were calculated to assess the taxa evenness and taxa richness for each sampling site. A combination of several diversity and biotic indices was calculated in order to take advantages of the strengths of each and develop a more complete understanding of community structure (Hooper et al., 2005).

Generalized linear models were developed in R software (version 3.0.3) (R Core Team, 2015), to investigate which variables determined the occurrence of water hyacinth. Prior to developing the GLM, correlations and variance inflation factors (VIFs) between the predictor variables were examined, in order to avoid problems of collinearity (Zuur et al., 2009). All variables with a correlation of 0.7 or higher were removed. The presence or absence of water hyacinth was the predicted variable, whereas temperature, conductivity, chlorophyll a, oxygen saturation and turbidity were retained as predictor variables. We started fitting the full model, i.e. including all candidate predictor variables. Next, a stepwise backward selection procedure was followed based on the Akaike Information Criterion (AIC), where the model with the lowest AIC value was retained as the final model. Relations between the residuals (the differences between observations and predictions by the retained model) and predictor variables were evaluated and the normality of the residuals was tested using a QQ-plot (probability plot). Retained models were only considered reliable if no relations between the residuals and the predictor variables were visually observed and residuals were normally distributed (Zuur et al., 2009); the retained models were rejected otherwise.

Non-parametric tests were performed, since the data were non-normal distributed (tested with Shapiro-Wilk Normality test). A Mann-Whitney U-test was used to compare physico-chemical variables between sampling sites with and without water hyacinth. Spearman’s rank correlation was used to explore the relationships between percentage water hyacinth cover and macroinvertebrate metrics. A Kruskal-Wallis ANalysis Of VAriance followed by post-hoc multiple comparisons was performed to test whether significant differences in ecological indices existed between different classes of vegetation cover. Species data were log(x+1) transformed before multivariate analysis to ensure data normality (Clarke, 1993). Non-metric Multidimensional Scaling (NMDS) was used to visualize the similarity of macroinvertebrate communities between habitatstypesonto two-dimensional charts. ANOSIM (ANalysis Of SIMilarities) was used for testing the similarities (based on the Bray-Curtis similarity) of macroinvertebrate community composition between sites with and without water hyacinth. A SIMPER (SIMilarity PERcentages) analysis was applied for identifying which species primarily contributedto the observed differences in macroinvertebrate assemblages between habitat types. All multivariate analyses were performed using the PRIMER software package (Clarke, 1993).

4.3 Results

4.3.1 Physico-chemical water quality

Table 4.1 shows the water quality results measured at the Daule-Peripa reservoir. Based on a Mann-Whitney U-test it was found that only turbidity differed significantly between sampling sites with and without water hyacinth (Table 4.1). Based on the low conductivity and high oxygen levels, a relatively good water quality was observed for most sampling sites, regardless of the presence or absence of water hyacinth.

Table 4.1 Median, minimum and maximum values of the measured environmental variables of the sampling sites with and without water hyacinth, statistical significance levels between both are indicated (Mann-Whitney U-test).

	Water hyacinth absent			Water hyacinth present			p-value
	Median	Min	Max	Median	Min	Max	
Temperature (°C)	28	26	29	27	26	30	0.63
Conductivity (µS/cm)	79	70	109	74	37	89	0.07
Total Dissolved Solids (mg/L)	0.1	0.1	0.1	0.1	0.1	0.1	0.20
pH	7	7	8	7	7	8	0.06
Chlorophyll a (µg/L)	7.0	5.5	25.4	6.9	4.7	10.1	0.61
Chlorides (mg/L)	2.2	1.7	2.5	2.5	1.3	2.9	0.10
Dissolved Oxygen (mg/L)	8	6	11	7	4	9	0.22
Oxygen Saturation (%)	102	81	131	90	52	118	0.22
Turbidity (FTU)	5	4	10	4	1	5	0.00

4.3.2 Habitat suitability of water hyacinth

After model selection only turbidity was retained in the final model (Appendix 4.1). Based on the generalized linear model we found that water hyacinth is present at sites with a low turbidity (Fig. 4.1). The Pearson correlation showed that there was a positive linear relationship between turbidity and chlorophyll a (r=0.65). Higher turbidity was observed at non-vegetated sites (Appendix 4.1, Fig. 4.2).

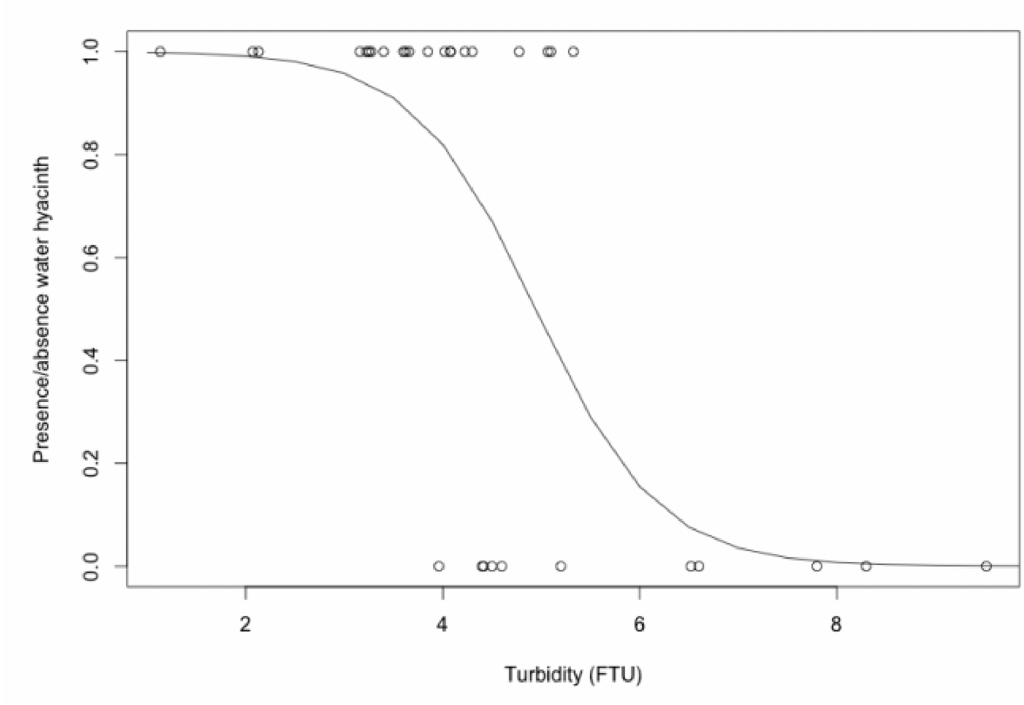

Figure 4.1 Generalized linear model for the presence/absence of water hyacinth in function of turbidity.

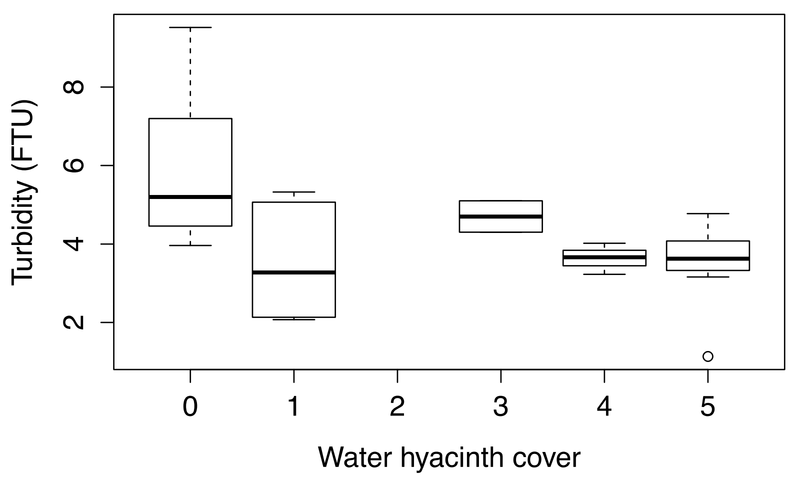

Figure 4.2 Turbidity in function of water hyacinth cover classes (0=absent, 1=1-5%, 2=5-25%, 3= 25-50%, 4 = 50-75%, 5 = 75-100%).

4.3.3 Relationship between the macroinvertebrate community and habitat characteristics

Based on the BMWP-Colombia scores the sampling sites of the Daule-Peripa reservoir were categorized in four water quality classes: moderate, poor, bad and very bad (Fig. 4.3). The BMWP-Colombia score and Margalef index were significantly higher (Mann-Whitney U-test, $p<0.01$) forsites containing water hyacinth compared to sites containing none. However, there were no distinct differences in the Shannon–Wiener index, Evenness index and Simpson index between the sampling sites with and without water hyacinth (all $p>0.05$). Figure 4.3 showed that, non-vegetated sites are close to each other and they are located near the dam. On the other hand, sites with high vegetation cover (class 5) are also close to each other, which are located far from the dam (Figure 4.3). Figure 4.6 showed that non-vegetated and high turbidiy sites are close to each other.

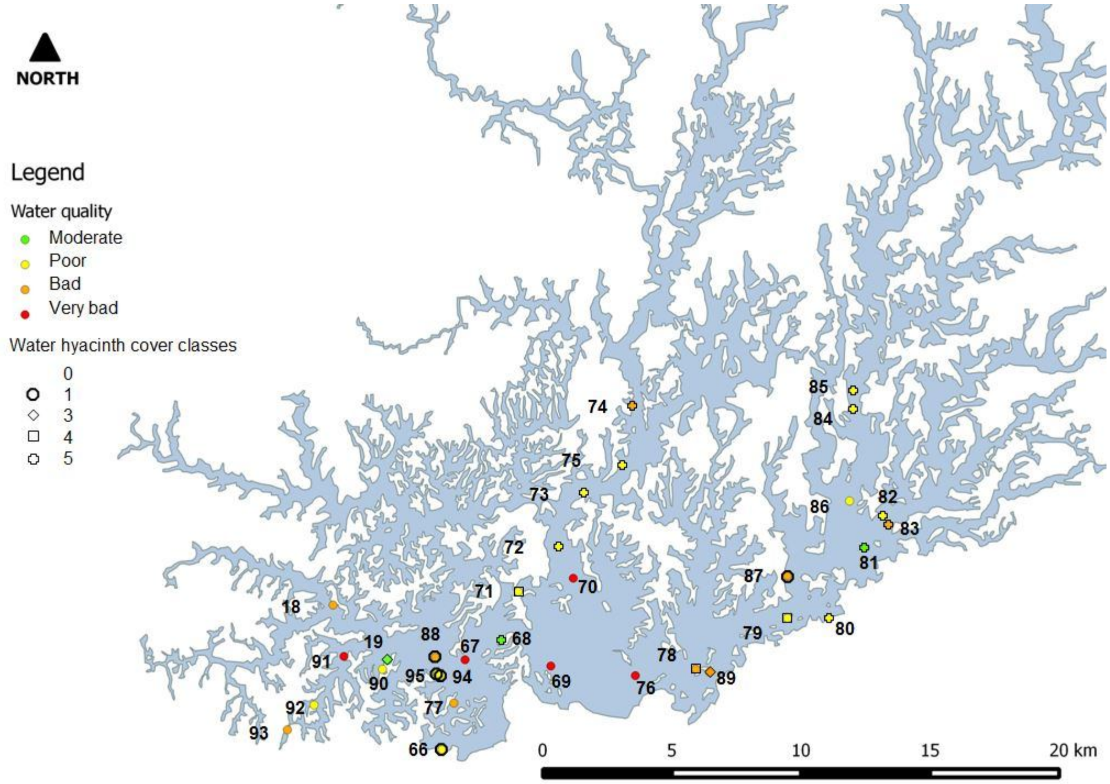

Figure 4.3 Geographic distribution of the ecological water quality in the Daule-Peripa reservoir based on the BMWP-Colombia score, with indication of the different vegetation cover classes of water hyacinth (numbers indicate sampling sites).

A Spearman rank-order correlation was run to determine the relationship between percentage of cover by water hyacinth and the ecological indices. There was a significant, positive correlation between percentage cover by water hyacinth and the BMWP-Colombia and Margalefindex (r = 0.64, r = 0.54 respectively; p < 0.05). Based on a Kruskal-Wallis ANOVA a significant difference was detected in BMWP-Colombia scores (chi-squared = 13.8, df = 4, p = 0.008) and in Margalef index between different vegetation classes (chi-squared = 12.6, df = 4, p = 0.01). Based on post-hoc tests we found a significant difference for the BMWP-Colombia index between vegetation class 0 and class 5 and for the Margalef index between vegetation class 3 and all other classes (p<0.05) (Fig. 4.4). When turbidity was plotted in function of the BMWP-Colombia, samples with a high percentage water hyacinth cover were clearly separated from sites without water hyacinth or sites with a low cover (Fig. 4.5). At non-vegetated sites, the turbidity was higher than 4 FTU and the BMWP-Colombia values were lower than 50 (water quality ranged from very bad to poor).

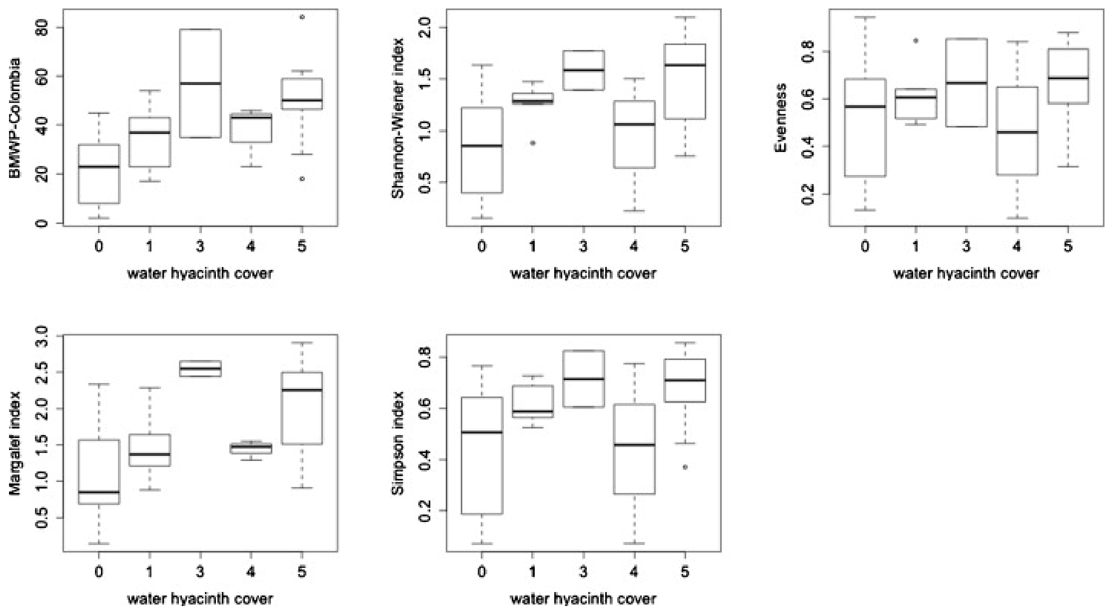

Figure 4.4 Values of BMWP-Colombia (a), Margalef index (b),Shannon-Wiener index (c), Simpson index (d) and Evenness (e) for different percentages of water hyacinth cover (0=absent, 1=1-5%, 3= 25-50%, 4 = 50-75%, 5 = 75-100%).

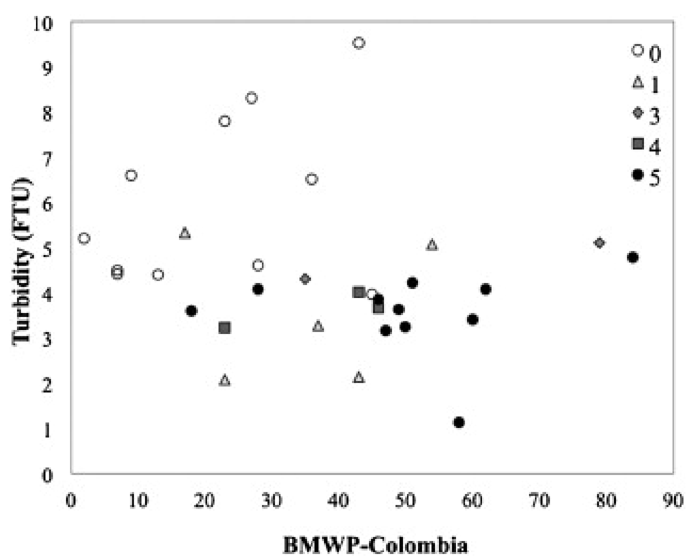

Figure 4.5 Scatter plot of the Biological Monitoring Working Party-Colombia (BMWP-Colombia) and the turbidity, with indication of the different vegetation cover classes of water hyacinth (0=absent, 1=1-5%, 3= 25-50%, 4 = 50-75%, 5 = 75-100%).

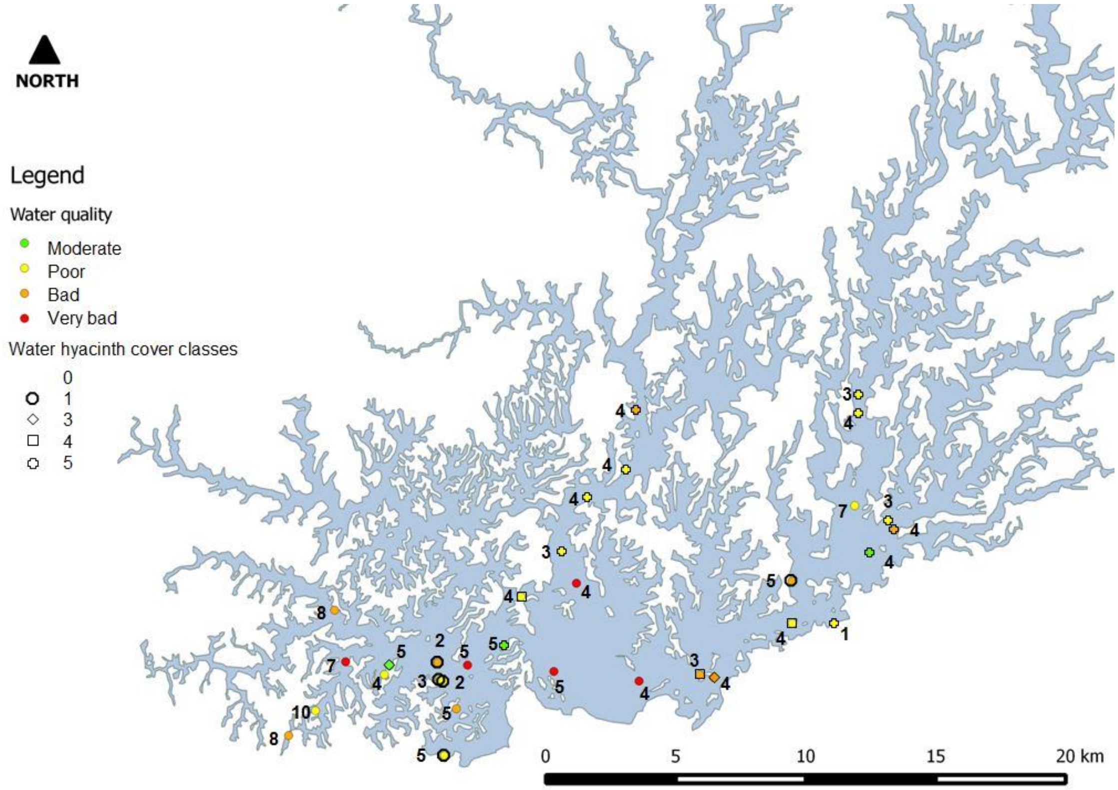

Figure 4.6. Geographic distribution of the ecological water quality in the Daule-Peripa reservoir based on the BMWP-Colombia score, with indication of the different vegetation cover classes of water hyacinth (numbers indicate turbidity values)

In total, 32 different macroinvertebrate taxa were found at the different sites with a maximum of 19 different taxa for sites without water hyacinth and a maximum of 25 taxa for sites covered by water hyacinth (Appendix 4.2). Sites with an intermediate vegetation cover (25-50%) had a relatively high abundance and diversity of macroinvertebrate taxa. SIMPER (SIMilarityPERcentages) analysis indicated that the similarity in species composition between samples with water hyacinth was 49.4%, while similarity between samples without water hyacinth was 29.6%. Dissimilarity in species composition between samples with and without water hyacinth was 67.6%. Densities of Chironomidae, Hyallelidae, Prostigmata, Dugesiidae, Libellulidae, Baetidae, Notonectidae, Coenagrionidae, Tubificidae, Mesoveliidae and Caenidae reached higher densities at sampling sites with water hyacinth, while Glossiphoniidae, Thiaridae, Gerridae and Corixidae reached higher densities at sampling sites without water hyacinth (Appendix 4.2). ANOSIM (ANalysis Of SIMilarity) indicated that there was a significant difference in macroinvertebrate community composition between sites with and without water hyacinth (R=0.47; p<0.01). Non-metric Multidimensional Scaling (NMDS) also indicated that samples with water hyacinth were more similar in macroinvertebrate composition to each other than samples without water hyacinth (Fig. 4.7).

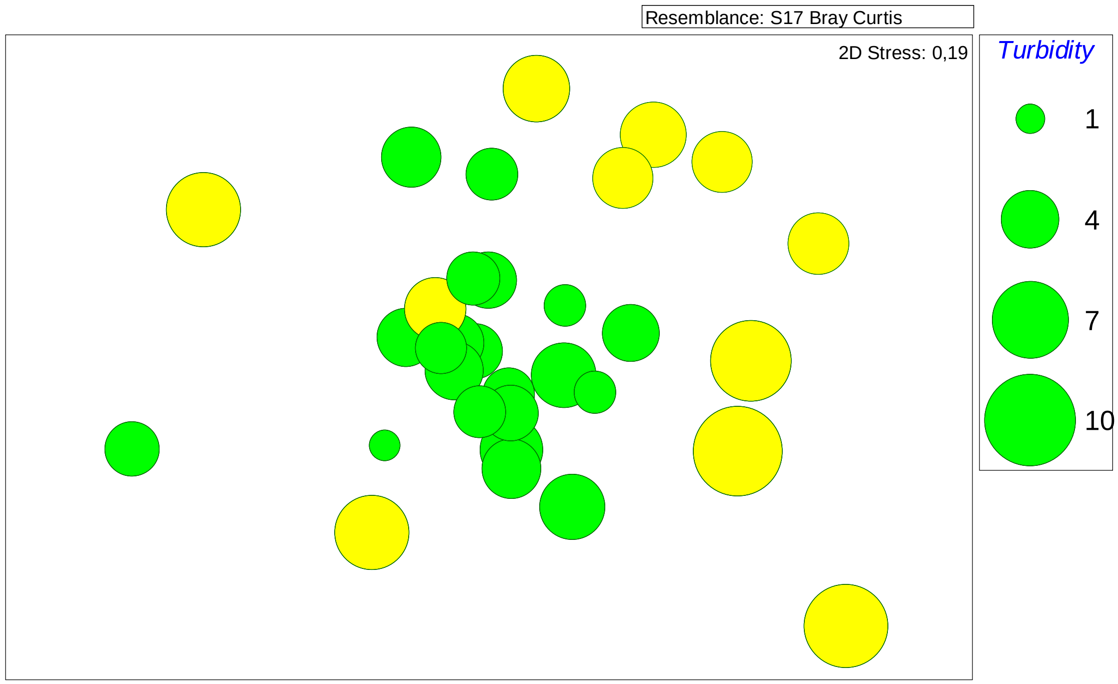

Figure 4.7 Non-metric Multidimensional Scaling (NMDS) plot with indication of samples with present (green) and absence (yellow) water hyacinth and turbidity.

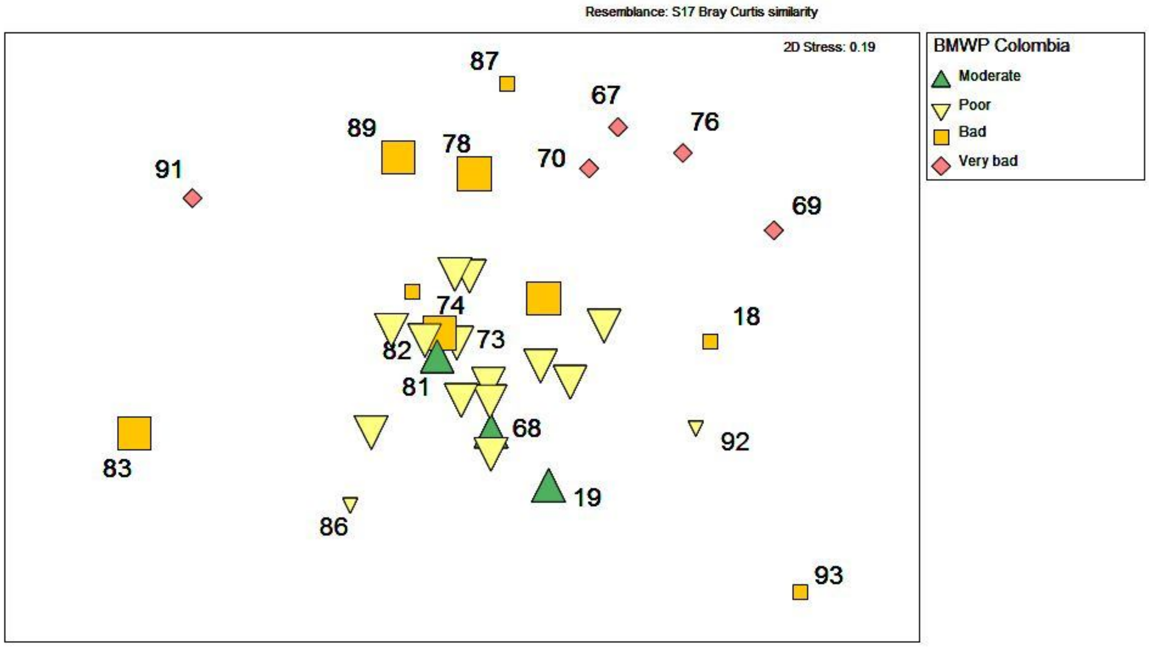

Figure 4.8 Non-metric Multidimensional Scaling (NMDS) plot with indication of samples with (large symbols) and without (small symbols) water hyacinth and the ecological water quality class according to the Biological Monitoring Working Party (BMWP) Colombia. Numbers indicate sampling sites which will be further analysed in taxa composition in the Table 4.2.

Table 4.2 showed that some higher BMWP-Colombia score taxa (e.g. Veliidae, Lymnaeidae, Naucoridae and Planorbidae) were alway presented at sites with water hyacinth.

Table 4.2: Macroinvertebrate composition at some sampling sites in the Daule-Peripa reservoir.

	BMWP score	Absence of water hyacinth										Presence of water hyacinth								
Taxa	Site	91	67	69	70	76	93	87	18	86	92	74	78	83	89	73	82	68	81	19
	Water quality	Very bad					Bad			Poor		Bad				Poor		Moderate		
Acari	-		p		p	p	p	p	p	p	p	p	p		p	p	p	p	p	p
Cambaridae	-		p							p						p				p
Dytiscidae	-																			p
Gerridae	-	p							p	p								p	p	
Tubificidae	1							p		p			p		p	p		p	p	
Chironomidae	2	p	p	p	p	p			p		p	p	p		p	p	p	p	p	p
Culicidae	2																p	p		
Hydrophilidae	3																	p		
Limoniidae	3										p									
Physidae	3																			p
Ceratopogonidae	5																			p
Corixidae	5						p				p									p
Glossiphoniidae	5			p	p	p	p	p	p	p	p							p	p	p
Libellulidae	5							p	p	p	p	p		p	p	p	p	p	p	p
Mesoveliidae	5																	p		p
Notonectidae	5						p			p	p						p	p		p
Thiaridae	5						p		p		p									p
Aeshnidae	6									p				p				p		p
Ampullariidae	6								p											p
Caenidae	6															p	p		p	p
Dugesiidae	6					p		p			p		p		p	p	p	p	p	
Baetidae	7						p			p	p	p	p		p	p	p	p	p	p
Coenagrionidae	7											p			p	p	p	p	p	p
Hyallelidae	7	p								p		p	p	p		p	p	p	p	p
Veliidae	7														p			p		
Lymnaeidae	8															p		p	p	
Naucoridae	8																		p	
Planorbidae	8																	p		

4.4 Discussion

This chapter is one of the few studies investigating the relationship between the occurrence of the invasive water hyacinth and physico-chemical water quality conditions and macroinvertebrate diversity in a tropical reservoir. In terms of physico-chemical characteristics, our results show that only turbidity differed significantly between sampling sites with and without water hyacinth. Water hyacinth reduces the effect of waves caused by the wind and motorboats, which explains the lower average turbidities that were observed at stations containing water hyacinth in the Daule-Peripa reservoir. On the other hand, water hyacinth will also more easily establish at sites where the wave action is not too strong, where there is a low flow velocity (c.f. chapter 5) and where consequently turbidity is low as well (Opande et al., 2004). Our habitat suitability model, indicating that water hyacinth is present at sites with a low turbidity, supports this finding. The Pearson correlation showed that there was a positive linear relationship between turbidity and chlorophyll (r = 0.65). It is possible that water hyacinth limits the development of phytoplankton via preventing light penetration and absorbing nutrients. In this way, water hyacinth reduces turbidity. Nevertheless, turbidity cannot be considered as a main cause for the presence of water hyacinth in Daule-Peripa reservoir, this relation should be rather considered as a correlation.

There was no significant difference in average chlorophyll a concentrations between covered and non-covered sites, however, the maximum concentration measured at water hyacinth absent sites was more than two times higher than that at sites where water hyacinth was present. This may be attributed to water hyacinth absorbing nutrients from the water column and reducing light penetration when forming dense vegetation mats (McVea and Boyd, 1975), thus limiting the productivity of phytoplankton at sites where water hyacinth is present (Villamagna and Murphy, 2010). Dissolved oxygen concentration, although not significantly different, was lower at sites where water hyacinth is present compared to sites where the species is absent. The lower oxygen levels measured, can be ascribed to the water hyacinth mats that prevent the transfer of oxygen from the air to the water and the reduced effects of mixing due to wind (Hunt and Christiansen, 2000). Since water hyacinth is a floating plant, the turbidity will not directly affect the growth, but water hyacinth growth can contribute to the clarity of the water by suppressing growth of algae due to light and nutrient competition, and moreover help to settle suspended solids due to the reduction of flows (e.g. induced via wind). Nevertheless, the cause-effect relationships between the presences of water hyacinth and other environmental variables was not investigated. Therefore, dedicated experiments need to be set up to have better understanding of this relation.

A positive relationship between the occurrence of water hyacinth and macroinvertebrate diversity was observed. Higher BMWP-Colombia and Margalef scores were reported for sites containing water hyacinth compared to locations without water hyacinth. The ecological water quality ranged from moderate to bad at sites with water hyacinth, while it ranged from poor to very bad at non-vegetated sites (Table 4.2). The Margalef index reveals that sites covered for 25-50% with water hyacinth have a higher species diversity compared to water hyacinth absent sites. Similarly, Kouame et al. (2011) found that the vegetation cover by water hyacinth positively altered the biodiversity of benthic invertebrate assemblages. Indeed, macrophytes can provide excellent microhabitats that promote the establishment and colonization of macroinvertebrates. Furthermore, macroinvertebrates use macrophytes for refuge and shelter against predation (Walker et al., 2013). In addition, the root mat of water hyacinth can provide new habitats for colonization by macroinvertebrates and in this way increase the diversity of aquatic macroinvertebrates (Masifwa et al., 2001; Rocha-Ramirez et al., 2007; Barker et al., 2014). When the percentage of water hyacinth cover was higher than 50% we did not observe an increase in positive effect, because the cover might be too dense, which negatively affects the physico-chemical water quality conditions and consequently also the diversity of macroinvertebrates. This may be explained by the intermediate cover hypothesis that was put forward by Villamagna (2009) to explain the observed patterns in invertebrate abundance and diversity of Lake Chapala in Mexico. This hypothesis suggests that the biotic community (e.g. fish) increases with increasing prey abundance and provision of refuge as a consequence of increased vegetation. However, a too high vegetation density may reduce oxygen availability and increase the competition between organisms thereby negatively affecting the density of the biotic community. Another aspect to consider is the indirect effect of water hyacinth cover on diversity and abundance of waterbird. Villamagna (2009) and Villamagna et al. (2012) found evidence for a more indirect influence on water birds via changes in trophic structure, prey community composition (e.g. macroinvertebrates) and energy flows throughout the system.

Our study showed that the dissimilarity in species composition between samples with and without water hyacinth was 67.6%, indicating differences in the taxa associated with water hyacinth. At sampling sites where water hyacinth was present, both pollution tolerant (e.g. Tubificidae and Chironomidae) and pollution sensitive (e.g. Hyallelidae, Baetidae and Coenagrionidae) families reached higher abundances compared to sites where water hyacinth was absent. In addition, some sensitive taxa (e.g. Veliidae, Lymnaeidae, Naucoridae and Planorbidae) are only presence at site with water hyacinth (Table 4.2). According to Rocha-Ramirez et al. (2007), the density of invertebrates is not only affected by the presence of water hyacinth, but also by environmental variables such as temperature, salinity, dissolved oxygen, and turbidity. Particular groups of invertebrates are correlated with specific variables (e.g. mayflies prefer a lower conductivity) (Rocha-Ramirez et al., 2007) and can respond differently to water hyacinth (Villamagna, 2009). In our study, it seems that the measured environmental conditions in combination with the presence of water hyacinth positively influenced the diversity and abundance of macroinvertebrates.

Water hyacinth is considered one of the world's worst weeds (Holm et al., 1977). The explosive growth of water hyacinth has caused serious socio-economic effects such as obstruction of water ways and reducing hydropower production (Masifwa et al., 2001). Our study showed that water hyacinth is positively related to macroinvertebrate diversity and water quality. This benefit should be weighed and compared to other impacts on ecosystem services before management actions are initiated. It was reported that about one third of the Daule-Peripa reservoir is covered by water hyacinth and that the presence of water hyacinth causes blockage of navigation between different communities in the reservoir and increases the incidence of infectious diseases (e.g. malaria, dengue and mosquito born infectious diseases) (Gerebizza, 2009). However, studies of water hyacinth effects on local economy (e.g. cost for removal of water hyacinth, cost of local people for transportation by boats, medical treatment and the problems with hydropower generation) in the Daule-Peripa reservoir have not been assessed. Small and isolated mats of water hyacinth can probably provide a unique habitat that contributes to overall biotic diversity and ecosystem functioning (Villamagna and Murphy, 2010). In case of the Daule-Peripa reservoir, we found that an intermediate vegetation cover of water hyacinth was positively related with the diversity of macroinvertebrates, whereas the impact of water hyacinth on hydropower generation is expected to be limited. However, further investigation is needed to quantify these findings and to assess the cost and benefits related to the presence of water hyacinth. Integrating the ecological knowledge obtained from this study with economic assessments and public perception could help decision makers to identify priority habitats to be targeted for the control of water hyacinth and to prioritize conservation actions in an operative way (Caplat and Coutts, 2011).

4.5 Conclusions

In conclusion, the findings of this chapter revealed that only turbidity was significantly different between sampling sites with and without water hyacinth. The presence of water hyacinth was an important variable affecting the diversity of macroinvertebrates in the Daule-Peripa reservoir. Despite the overall positive relationship between macroinvertebrate diversity and the occurrence of water hyacinth, it is important to consider the socio-economic costs related to the management of water hyacinth. This work is believed to serve as baseline data for further studies on tropical reservoirs and as an incentive to assess the indirect effect of water hyacinth on other communities (e.g. fish, phytoplankton, zooplankton and waterbird).

Chapter 5: Threshold responses of macroinvertebrate communities to stream velocity: a case study from the Guayas River basin in Ecuador

Adapted from:

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Chapter 5: Threshold responses of macroinvertebrate communities to stream velocity: a case study from the Guayas River basin in Ecuador

Abstract

The Guayas River basin is one of the most important water resources in Ecuador, but the expansion of human activities has led to a degraded water quality. The purpose of this study was: (1) to explore the importance of physical-chemical variables in structuring the macroinvertebrate communities and (2) to determine if the thresholds in stream velocity related to macroinvertebrate community composition could be identified in the Guayas River basin. Macroinvertebrates and physical-chemical water quality variables were sampled at 120 locations during the dry season of 2013 in the Guayas River basin. Canonical Correspondence Analysis (CCA) was performed to identify relevant physical-chemical characteristics of the river influencing the distribution of the macroinvertebrate communities. According to the CCA, the most important environmental factors influencing the distribution of macroinvertebrate communities are stream velocity, chlorophyll a, conductivity, temperature and elevation. Threshold Indicator Taxa ANalysis (TITAN) was able to discriminate between the macroinvertebrate communities related to stagnant water (Daule-Peripa reservoir) and the macroinvertebrate community related to running waters. The results provide essential information to further support water management plans of the Guayas River basin. Information obtained will be useful for management of similar rivers in South-America, as well as the rest of the world.

5.1 Introduction

Rivers are ecosystems, which provide great ecological value (Benetti et al., 2012). They are also an important source of renewable water supply for humans and freshwater ecosystems (Vorosmarty et al., 2010) and provide many ecosystem services such as sources of drinking water and recreational areas and provide nursing grounds and food for many organisms (Berger et al., 2016). However, the increase of human activities such as industrialization, urbanization and intensive agriculture cause river degradation (Carpenter et al., 2011). Freshwater organisms are impacted via various stressors, such as water pollution, erosion, alterations in stream hydrology and changing habitat structures (Allan, 2004). It is estimated that at least 10,000–20,000 freshwater species are extinct or at risk of extinction (Vorosmarty et al., 2010). The water needs for human and natural ecosystems are often considered as competing with each other (Richter et al., 2003). At the same time, water managers and political leaders need to manage water to meet human requirements, to protect endangered species and to support freshwater ecosystems (Richter et al., 2003).

The Guayas river basin is one of the most important river systems in Ecuador (Arias-Hidalgo, 2012) and provides a high contribution to Ecuador's gross domestic income (Andres, 2009). The Guayas basin is facing many water resources problems such as increasing modification of the natural flows by dams and water extractions for agriculture and urban water supply (Waite, 1982). The intensive use of natural resources in the Guayas basin leads to exhaustion and disequilibrium of the ecosystem and the ecological integrity (Madonado, 2011). Because of the Daule-Peripa hydropower dam, the discharge in the Daule River is highly variable. We expected that stream velocity plays a strong role affecting the macroinvertebrate community. However, the macroinvertebrate community of the Guayas River basin has been little studied and research investigating the tipping point where species are likely to be affected by changing stream velocity is lacking. There is a strong need for information about how macroinvertebrate communities are affected by environmental stressors in the Guayas river basin in order to support the management for conservation and restoration of aquatic systems. Therefore, the aim of this study was to explore which physical-chemical water quality variables are most important in structuring the macroinvertebrate communities. Furthermore, we checked whether thresholds in stream velocity related to macroinvertebrate community composition could be identified. These threshold values are needed in order to inform policy makers about critical levels of stream velocity, which can cause rapid changes in macroinvertebrate communities. The results provide useful information for prioritizing management actions in the Guayas river basin and it will be particularly useful for management of similar rivers in South-America, as well as the rest of the world.

5.2 Data analysis

All statistical analyses, including data exploration and model development were done using R software (version 3.2.3) (R Core Team, 2015). A protocol for data exploration as described by Zuur et al. (2010) was used to avoid common statistical problems. Prior to the actual data analysis, the initial data set was tested for outliers and normality. The boxplots were made to visualize the variability of all measured variables (Appendix 5.1). The Shapiro-Wilk test was used to check the normality of the data. As the data were non-normal distributed, the Spearman's correlation coefficient was used to investigate correlations between physical-chemical variables. Following the procedure suggested by Zuur et al. (2010), TDS and pH were excluded from further analysis because they were highly correlated to conductivity and DO (r = 0.95, r = 0.75, respectively) and to avoid problems of collinearity (Appendix 5.2).

Kruskal–Wallis tests followed by Dunn test post hoc multiple comparisons were performed to test whether significant differences existed between the five ecological water quality classes for elevation, temperature, conductivity, chlorophyll a and stream velocity. The Dunn test was used because there was unequal number of sampling sites in different water quality classes (Zar, 2010). The Dunn test was performed using the *DunnTest* function in the *DescTools* package (Signorell, 2016) in R software (version R.3.2.3). All tests were determined at the 5% significance level.

Detrended Correspondence Analysis (DCA) was performed to test the appropriate response model of the macroinvertebrate metrics to the environmental data. The first axis of the DCA gradient length was 4.71, which is more than three standard deviations, therefore, the unimodal ordination method was used as recommended by Šmilauer & Lepš (2014). A unimodal relationship was expected for macroinvertebrate taxa along the gradient defined by the explanatory environmental variables. The Canonical Correspondence Analysis (CCA) is a constrained ordination technique, in which the response variable set is constrained by the set of explanatory variables (Paliy and Shankar, 2016). Thus, CCA was developed to explore which environmental variables could be important in structuring the macroinvertebrate communities in the Guayas River basin. One sampling location, where no macroinvertebrates were found, was excluded from DCA and CCA analysis. As the data were non-normal distributed, all data were $\log_{10}(x + 1)$ transformed prior to the DCA and CCA analysis for normalization. DCA and CCA were performed using the *vegan* package (Oksanen et al., 2016) in R software (version R.3.2.3).

Threshold Indicator Taxa ANalysis (TITAN) was used to detect community responses to stream velocity in the Guayas River basin. In total, 83 macroinvertebrate taxa were found in 120 sampling sites within the Guayas river basin. Among these, 54 had five or more occurrences and were included in the TITAN analysis (Baker et al., 2015). Taxa names which have more than eight characters were coded as eight character abbreviations of scientific names. Stream velocity was used as a predictor variable. Abundance data were not transformed because transformation is unnecessary in TITAN 2. Bootstrapping (1000 repetitions) was implemented to estimate uncertainties around change point locations and response direction (positive or negative) of threshold indicator taxa. 1000 permutations were used to determine species specific z scores. Multiple permutations (e.g. 500 or 1000) are recommended in order to have more precise z scores (Baker et al., 2015). We set as a requirement that the minimum number of observations on either side of any partition had to be ≥ 3. TITAN was performed in the package TITAN 2 (Baker et al., 2015) in R software (version R.3.2.3). Details of the TITAN method can be found in Baker and King (2010), King and Baker (2014) and Baker et al. (2015).

5.3 Results

5.3.1 Relationship between physical-chemical conditions and macroinvertebrate communities

Table 5.1 summarizes the physical-chemical water quality measured at 120 sampling locations within the Guayas river basin. In addition, the boxplots refect the variabilities of all measured variables are presented in the Appendix 5.1. The water was stagnant at all sampling sites in the Daule- Peripa reservoir. There was one sampling site located at the tributary of the Daule River, which had the highest observed values of DO (14 mg/L), chlorophyll a (67 µg/L), chlorides (167 mg/L) and conductivity (1981 µS/cm). Conductivity values measured at all sampling sites at the Daule-Peripa reservoir were less than 110 µS/cm.

Table 5.1 Median, mean, maximum, minimum values and standard deviation of continuous environmental variables measured in the Guayas river basin

Variable	Median	Mean	Max.	Min.	Std.
Temperature (˚C)	26	26	34	19	2.5
pH	7	7	9	6	0.5
Dissolved Oxygen (mg/L)	8	8	14	2	1.7
Chlorophyll a (µg/L)	3	6	67	0.7	8.7
Chloride (mg/L),	2	7	182	0.5	22.8
Turbidity (FTU)	3	10	356	0	35.1
Conductivity (µS/cm)	123	200	1981	36	238
Total Dissolved Solids (TDS)	0.0	0.1	1.3	0.0	0.15
Elevation (m a.s.l.)	82	135	1075	2	187
Velocity (m/s)	0.1	0.2	1.5	0.0	0.3

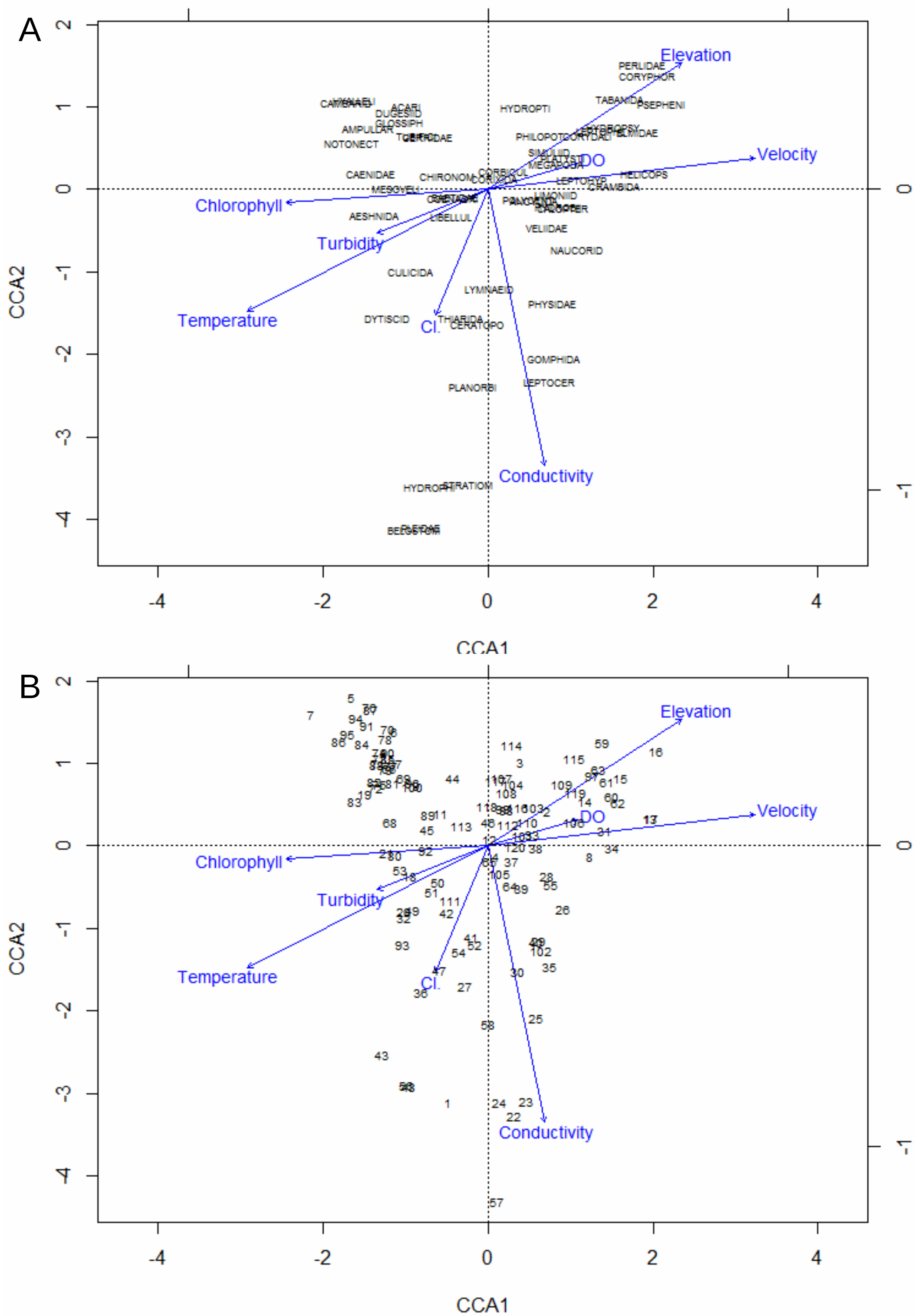

Figure 5.1 Canonical Correspondence Analysis (CCA) diagram showing the different macroinvertebrate taxa (A) or stations (B) and their correlation with environmental variables (blue vectors). Letters indicate taxa, numbers indicate sampling sites.

CCA analysis indicated an association between macroinvertebrate taxa and chlorophyll a, conductivity, temperature, elevation and stream velocity. The first two ordination axes of the CCA analysis explained 45% and 22% of the total variance, respectively (Fig. 5.1, Appendix 5.4). The first axis was positively correlated with velocity and elevation, while negatively correlated with temperature and chlorophyll a. However, the correlations are strong for velocity and temperature and weak relations with elevation and chlorophyll a (Appendix 5.4). Regarding the first axis, sampling sites with a high elevation and high velocity are located at higher altitude areas with fewer anthropogenic disturbances. On the other hand, sampling sites by increased levels of chlorophyll, conductivity and temperature were collected at the downstream part of both the Daule and Babahoyo River (Fig. 5.1). The second axis was strongly negatively correlated with conductivity (Appendix 5.4). Conductivity and chlorophyll a might have strong impacts at some particular sites. In general, stream velocity is key variable influencing the distribution of the macroinvertebrate communities.

Based on the BMWP-Colombia scores, the sampling sites within the Guayas River basin were categorized into five water quality classes: good, moderate, poor, bad and very bad (Fig.5.2). The result of the Kruskal–Wallis test shows that there was no significant difference in conductivity between different water quality classes ($p>0.05$). However, there were significant differences in elevation, temperature, chlorophyll a and stream velocity between different water quality classes ($p<0.05$). Good water quality was observed for sampling sites characterized by high elevations, high stream velocities, low temperatures and low levels of chlorophyll a ($p<0.05$) (Fig. 5.3).

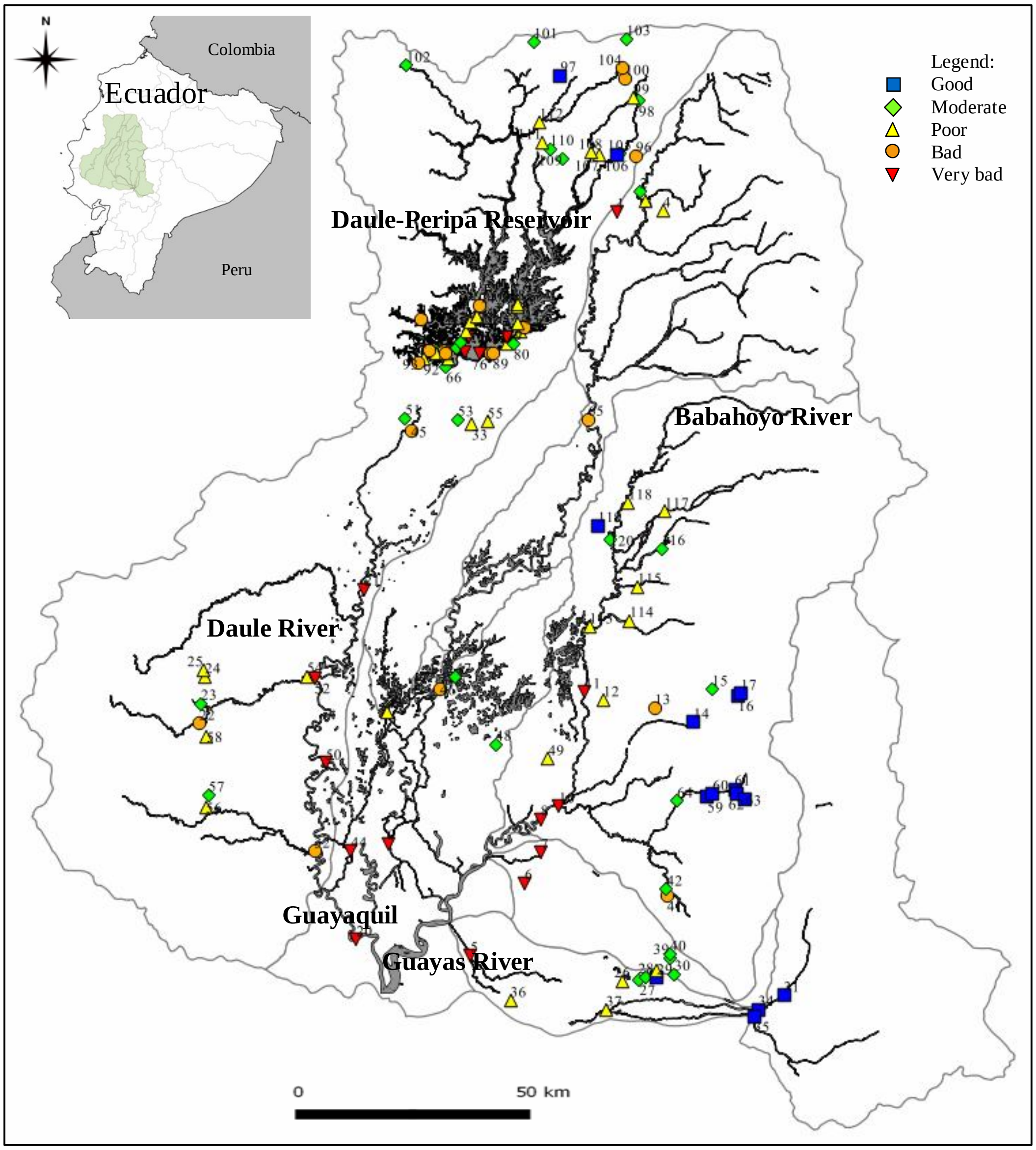

Figure 5.2 Map of the study area in the Guayas river basin with indication of the ecological water quality based on the BMWP-Colombia for each sampling site

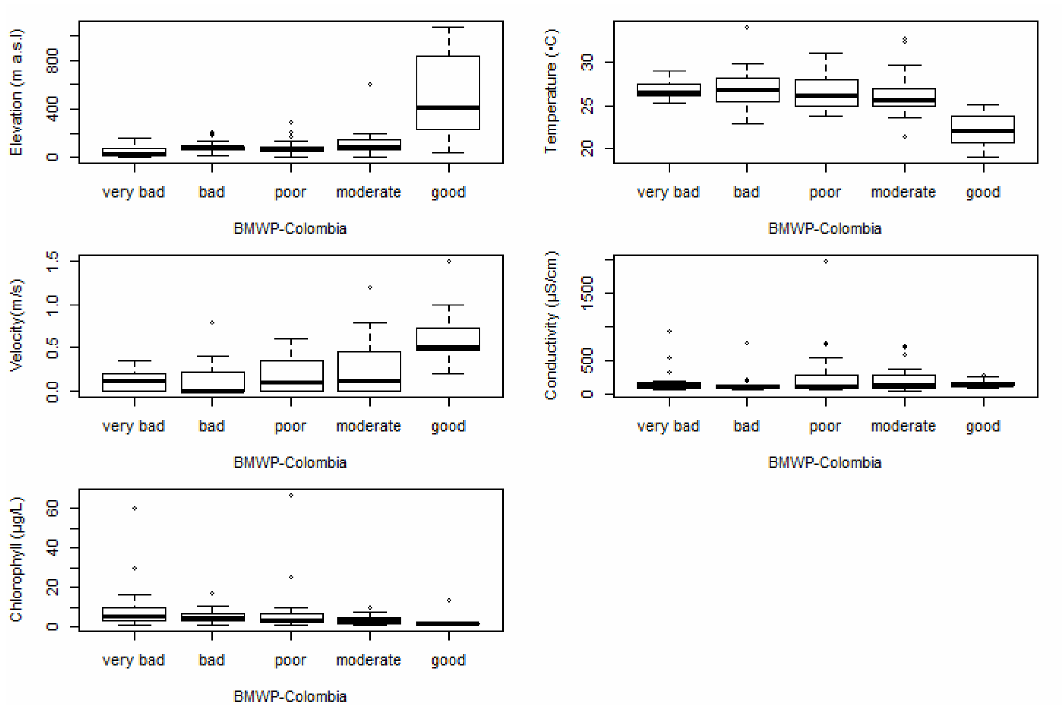

Figure 5.3 Boxplots of the different ecological water quality classes for the main environmental variables (elevation, temperature, velocity, conductivity and chlorophyll a) in the Guayas River basin. Bold horizontal lines represent median, boxes represent interquartile ranges (25–75% percentiles) and range bars show maximum and minimum values, small black squares show outliers

When the presence/absence of water hyacinth was plotted in function of flow velocity, the result indicated that the water hyacinth was only presence at sites with flow velocity lower than 0.4 m/s (Figure 5.4). Moreover, when water hyacinth cover classes were plotted in function of the flow velocity, the result showed that the percentage cover of water hyacinth was higher (class 4 and class 5, >50%) at sites characterized by a flow velocity lower than 0.1 m/s (Figure 5.5).

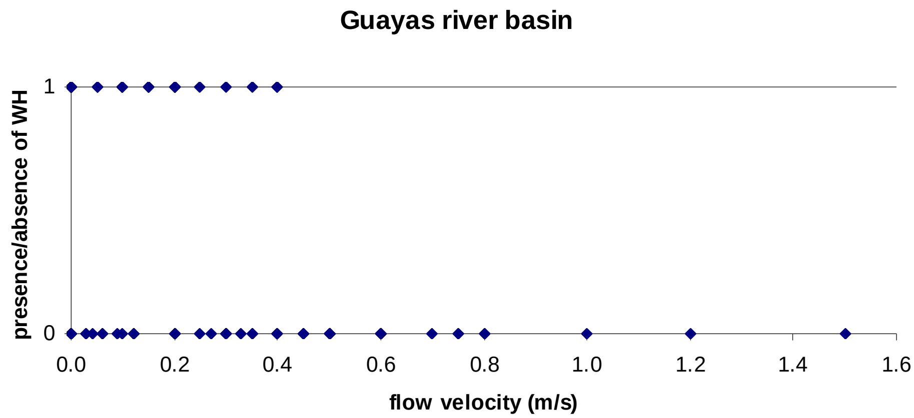

Figure 5.4. Plot illustrating the presence/absence of water hyacinth in the function of flow velocity.

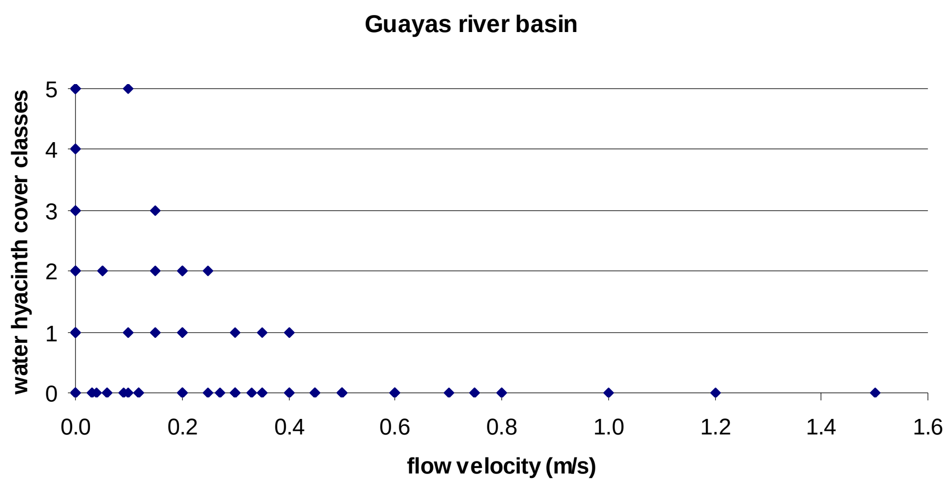

Figure 5.5. Plot with water hyacinth cover class in the function of flow velocity

5.3.2 Threshold change points and indicator taxa

Threshold Indicator Taxa ANalysis was implemented to test the macroinvertebrate community response to stream velocity in the Guayas river basin. The result of TITAN showed clear tipping points for the thresholds defined by stream velocity at 0.03 m/s and 0.4 m/s. TITAN revealed 34 taxa (63%) as the reliable indicators of stream velocity, of which 20 taxa are indicated as indicators of high stream velocity and 14 taxa as indicators of low stream velocity in the Guayas river basin (Fig. 5.6).

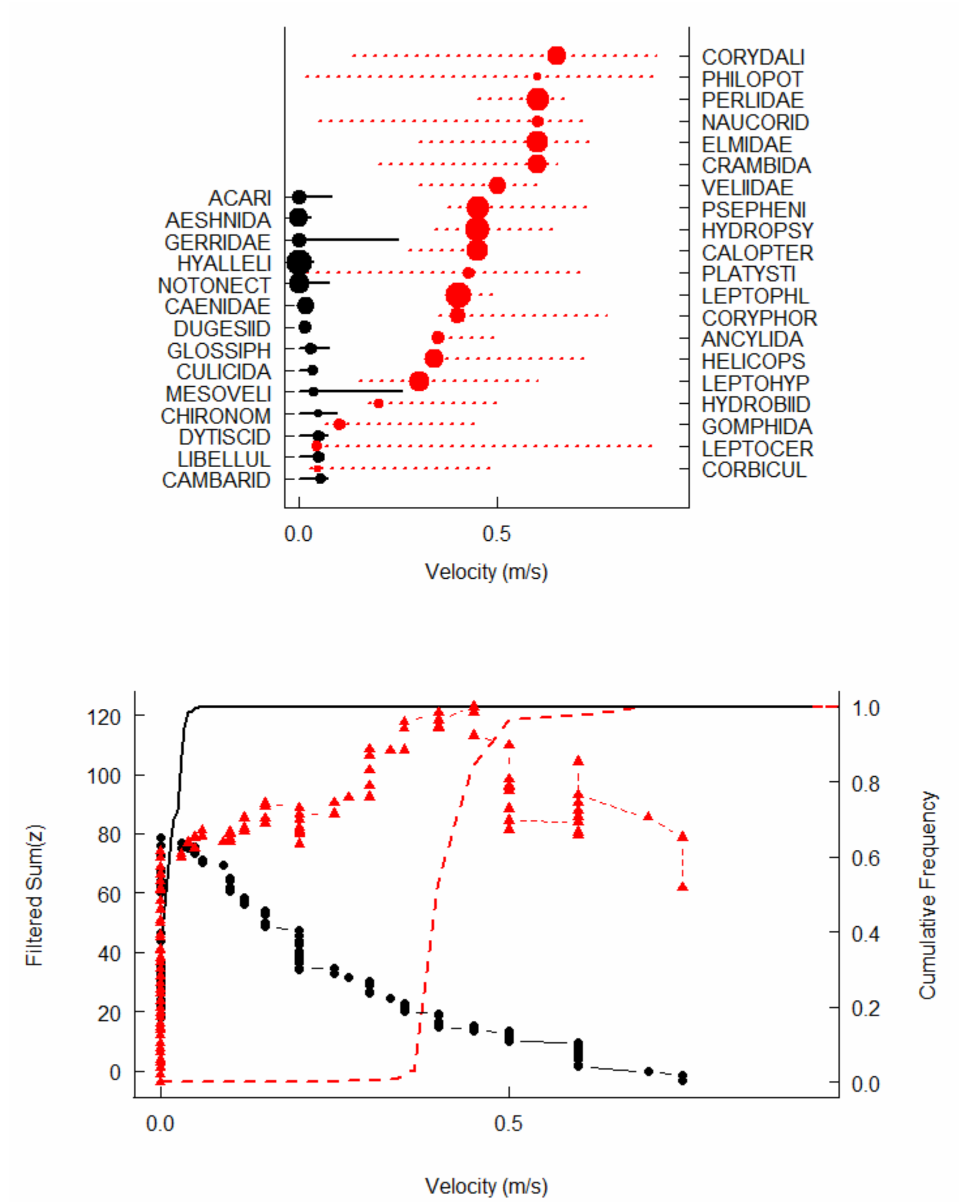

Figure 5.6 Threshold Indicator Taxa Analysis and change points (dots) for macroinvertebrate community response to stream velocity (p ≤0.05, purity = 0.95, reliability = 0.95 for 5 minimum number of observations, 1000 bootstrap and 1000 permutation replicates). Negative indicator taxa (z–) are indicated by black symbols and lines and positive indicator taxa (z+) are indicated by red symbols and dashed lines. Solid and dashed lines are cumulative frequency distributions of sum(z-) and sum(z+) maxima (respectively) across bootstrap replicates. The size of change point symbol (dots) is proportional to the magnitude of the taxa response. Z− species responded negatively to an increase in stream velocity, Z+ species responded positively to an increase in stream velocity. Horizontal lines suggest 5-95% quantiles from the bootstrapped change point distribution. Abbreviations and tabular results for individual taxa are found in the Appendix 5.3.

5.4. Discussion

5.4.1 Environmental influences on macroinvertebrate community

According to the CCA results, there was a strong association between macroinvertebrate taxa and chlorophyll a, conductivity, stream velocity, temperature and elevation. This suggests an important linkage between physical-chemical variables and macroinvertebrate community in the Guayas River basin. Our results show that good water quality was observed at sampling sites at high elevation, low temperature, high stream velocity and low levels of chlorophyll a. Those characteristics were observed at upstream sites, which have fewer human impacts. In general, water quality of rivers often deteriorates as one moves downstream due to the accumulating effects of different types of anthropogenic activities (Selvanayagam and Abril, 2015). Low ecological water quality was observed at sites with high chlorophyll a concentrations and high conductivity. The few sampling sites at small tributaries of both the Daule and Babahoyo Rivers and which were almost dry during the sampling, were characterized by a high conductivity. In addition, the distribution of data of the Guayas river basin (Appendix 5.1) showed that the conductivity has a median value = 123 µS/cm, 75th Percentile = 192 µS/cm; chlorophyll has a median value = 3.1 µg/l, 75th Percentile = 6.7 µg/l, velocity has a median value = 0.1 m/s and 75th percentile = 0.4 m/s. Thus conductivity and chlorophyll might affect the invertebrates strongly at some particular sites, but stream velocity is key variable influencing the distribution of the macroinvertebrate communities in whole Guayas river basin. Water extraction and irregular water release from Daule-Peripa dam caused low water levels at some sampling sites. It is possible that conductivity and chlorophyll reached to high values because of high concentration effects and increased residence time. In this way, damming, water extraction, intensive agriculture and urbanization can generate negative effects to water quality in the Guayas River basin.

The Daule-Peripa reservoir receives water from the Daule and Peripa Rivers (Gelati et al., 2011). The water depth of the reservoir fluctuates between 70 and 85 m due to the operation of the dam, while the natural water depth of the reservoir ranges between 79 to 81 m due to changes in rainfall (CELEC, 2013). Dams cause fluctuations in flow rate and accumulation of fine sediment within the reservoir (Käiro et al., 2011). Stream velocity and sediment substrate affect taxa distribution, abundance, richness and diversity (Alvarez-Mieles et al., 2013), especially sensitive macroinvertebrate taxa (Timm et al., 2011). It has been shown that changes in the flow regime from water regulation and desiccation cause a significant reduction in macroinvertebrate density (Theodoropoulos et al., 2015). The results of this research were supported by previous relevant studies carried out from different locations with varying climatic and habitat conditions. For examples, in Malaysian streams, stream velocity is one of the most important variables, which influences macroinvertebrate diversity (Rawi et al., 2013). The case study in the Sacramento River in California, United State, where dams control water for flood prevention, hydropower and irrigation purposes revealed that current velocity was the most important factor explaining community composition of macroinvertebrates (Nelson and Lieberman, 2002). In Kangaroo River (Australia), velocity was the most important characteristic affecting the spatial pattern of macroinvertebrate abundance and diversity (Brooks et al., 2005). The Daule-Peripa dam is characterized by stagnant water and there might be a high volume of sediment accumulation, which can affect the macroinvertebrate community.

Our study showed that sites with a good water quality has a median value of chlorophyll a of 1.7 µg/L, a conductivity of 126 µS/cm and stream velocity of 0.5 m/s. Algae are good bioindicators for eutrophication (Hausmann et al., 2016). High chlorophyll a concentrations indicate the occurrence of algal blooms (Huanga et al., 2011). The growth of algae can provide evidence of eutrophication and degradation of water quality in streams and rivers (Porter et al., 2008). In general, the conductivity values at sampling sites within the Guayas River basin are quite low (median = 123 µS/cm), except some locations that were almost dry. Low rainfalls during the dry season decrease the water level and as a consequence, possibly indicating severe water quality problems (little water remained and high conductivity values (maximum value was 1981 µS/cm) at some sampling sites. The Guayas river basin covers nine provinces with a total population of 4.8 million (Arias-Hidalgo, 2012). The main economic sectors of the Guayas basin are hydropower, fisheries and agriculture (e.g. banana, rice, maize, African oil palm *Elaeis guineensis* and cacao) (Alvarez-Mieles et al., 2013). The anthropogenic activities are key issues, which can lead to drastic changes in aquatic ecosystems and which can heavily impact the water quality of this river basin (Forio et al., 2015; Damanik-Ambarita et al., 2016b). The study in the United State showed that river flow fluctuations associated with dam operations, water pollution and invasive species have been identified as three leading causes of the imperilment of aquatic animals (Richter et al., 1997).

5.4.2 The threshold responses of the macroinvertebrate community to stream velocity

In this study, the TITAN identified the tipping points for the thresholds defined by stream velocity at 0.03 m/s and 0.4 m/s. The TITAN method was only developed in 2010 and therefore, only a few studies used TITAN to detect thresholds for changes in macroinvertebrate community composition. We did not find other studies related to thresholds for stream velocity, thus, this research is probably the first that identified threshold responses of macroinvertebrates to stream velocity using Threshold Indicator Taxa Analysis. The results obtained during the present study are in accordance with previous studies. According to the study conducted at the upper, mountainous reaches of four rivers in central Greece, the Boosted Regression Tree models revealed that high macroinvertebrate abundance and diversities were detected in flow velocities between 0.3 m/s and 0.75 m/s (Theodoropoulos et al., 2017). The habitat suitability model indicates the suitable current velocity for certain macroinvertebrate taxa such as *Baetis* ranges within 0.3–0.7 m/s, and the optimum current velocity is 0.4 m/s (Li et al., 2009). Furthermore, it was reported that at a stream velocity higher than 1 m/s, velocity acts as a constraint for most living organisms and the habitat is colonized only by species that tolerate such a high velocity (Wang and Xu, 2012).

TITAN revealed 34 taxa (63%) as the reliable indicators of stream velocity, of which 20 taxa are associated with a high stream velocity and 14 taxa are associated with a low stream velocity in the Guayas river basin. Everaert et al (2014) found that an increase in stream velocity was correlated with an increasing probability of occurrence of Leptophlebiidae in the Chaguana river basins in Ecuador. Our findings are in line with previous studies that the Leptohyphidae were frequently living at stream velocities ranging from 0.37 to 0.64 m/s and that increasing stream velocity is correlated to increased abundance of mayflies Leptophlebiidae in the Bento Gomes River, Brazil (Nolte et al., 1997). We also found that high stream velocities are associated with numerous sensitive taxa such as Perlidae and Psephenidae (tolerant scores 10), Leptoceridae (tolerant score 8) and Hydropsychidae (tolerant score 7). These results provide evidence for the possible influence of stream velocity on the macroinvertebrate community and the water quality.

TITAN revealed that 14 taxa such as Caenidae, Hyalellidae, Notonectidae, Dugesiidae and Glossiphoniidae displayed a negative response to an increasing stream velocity. When examining the habitat preferences for those taxa, the majority is strongly associated with vegetation (e.g. Notonectidae, Hyalellidae, Glossiphoniidae, Mesoveliidae, Aeshnidae, Libellulidae), bottom sediments (e.g. Hyalellidae) and shallow lakes (e.g. Dugesiidae, Gerridae, Mesoveliidae, Dytiscidae). Caenidae are dominant in stagnant water (Nolte et al., 1997). Nelson and Lieberman (2002) found the highest abundance of Chironomidae was observed at low velocities. High flow velocity is associated with the absence of water hyacinth. Therefore, this finding may be explained by a combination of unfavorable stream velocity and the absence of water hyacinth, which negatively affects the abundance and frequency of occurrence of the negatively associated taxa. It is possible that taxa which did not respond to environmental variables are more tolerant, or they are able to use different types of habitat. Moreover, case studies from Neotropical region (e.g. Bento Gomes River, Brazil) emphasized the seasonal discharge variation and velocity fluctuations affect macroinvertebrate community (e.g. mayfly) assemblages (Nolte et al., 1997). Therefore, study over the wet and dry season is needed to have full understanding of how the seasonal stream velocity differences affect the macroinvertebrate community in the Guayas River basin.

TITAN results provide information of the habitat preference (stream velocity) of macroinvertebrates and the evidence of local community shifts due to the change of stream velocity in the Guayas river basin. In other words, TITAN indicates the change of macroinvertebrate community due to the installation and operation of hydropower dams related to overall impacts on the flow. The sensitive taxa like Leptophlebiidae will increase in abundance and frequency of occurrence due to the increase of stream velocity. In this way, the management of flow of the dam (translated to flow velocity) can assist sensitive groups in increasing abundance and frequency by a proper operation. Moreover, the change point values can be used for detecting ecological flow. For example, the threshold for positively associated taxa (sensitive taxa) = 0.4 m/s, thus, a hydraulic model can be used to quantify how much water should be discharged. Nevertheless, the obtained tipping-point values should be considered as primary results as this is only the first analysis and publication on macroinvertebrate communities in the Guayas River and therefore future validation is needed based on additional data.

5.5 Conclusions

In conclusion, this chapter provides an understanding of the relationships between macroinvertebrates and environmental characteristics in the Guayas river basin. Chlorophyll a, conductivity and stream velocity are influential variables that need to be considered as important ecological drivers for macroinvertebrate communities. The results provide clear tipping points in stream velocity. TITAN was able to discriminate between the macroinvertebrate community related to stagnant water (Daule-Peripa reservoir) and the macroinvertebrate community related to running waters. The results indicate the importance of multiple factor assessments for reliable predictions of macroinvertebrate responses.

Chapter 6: Use of Threshold Indicator Taxa ANalysis (TITAN) to detect macroinvertebrate community responses to environmental gradients in the Portoviejo River (Ecuador)

Adapted from:

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Chapter 6: Use of Threshold Indicator Taxa ANalysis (TITAN) to detect macroinvertebrate community responses to environmental gradients in the Portoviejo River (Ecuador)

Abstract

The Portoviejo River, located in the central western part of Ecuador, has been heavily impacted by human activities. Information on water quality is indispensable for sustainable decision-making and water management. The aims of this chapter were to assess the water quality, to analyze the change of community structure along the key environmental variables and to find potential indicator taxa of macroinvertebrate communities in the Portoviejo River. In this chapter, macroinvertebrates and physico-chemical variables were sampled and hydro-morphological conditions were recorded at 31 locations during the dry season of 2015 in the Portoviejo River. The Biological Monitoring Working Party-Colombia (BMWP-Colombia) values were calculated to assess the ecological water quality. The BMWP-Colombia scores indicated that water quality of the sampling sites within the Portoviejo River ranged from good to bad. Threshold Indicator Taxa ANalysis (TITAN) was used to examine changes in macroinvertebrate communities along environmental variables. TITAN revealed clear tipping points in elevation, conductivity and nitrate-nitrogen concentrations affecting the macroinvertebrate community. Atyidae, Corbiculidae, Thiaridae, Acari, Baetidae and Leptohyphidae can be considered as indicator taxa to assist in identifying environmental thresholds of the Portoviejo River. This chapter suggests that conductivity and nitrate-nitrogen concentrations are influential variables, which need to be considered as important ecological drivers for the conservation efforts in the Portoviejo river, in particular related to the future investment in wastewater treatment. This approach could be applied as a useful management tool to support future management of other similar rivers, and determine protection and restoration actions.

6.1 Introduction

Rivers are one of the most important freshwater resources for human life (Chapman, 1996). Rivers provide many ecosystem services such as source of drinking water, irrigation of croplands, industrial and municipal water supply, waste disposal, fishing, sightseeing, shipping and aesthetic value (Chapman, 1996; Pan et al., 2012). However, rivers are highly vulnerable to anthropogenic disturbance (e.g. urbanization, changes in land use, intensification of agriculture) (Bredenhand and Samways, 2008). The increase in population and human activities often lead to habitat degradation, poor water quality (Kibena et al., 2014) and reduced ecosystem services (Pan et al., 2012). River systems are affected severely by natural flow manipulation, altered water temperatures, river channel modification, floodplain transformation and the disruption of river continuity. Human disturbance, for example, the construction of dams in the river leads to a change of the hydrological conditions, modification of the flow regime and sediment transportation (Takao et al., 2008), thus influences aquatic ecosystems strongly (Zhang et al., 2010).

Freshwater organisms are impacted via various stressors, such as water pollution, erosion, and alterations in stream hydrology and changing habitat structure (Allan, 2004). Aquatic macroinvertebrates have been used for freshwater monitoring and assessment for several decades (Smith et al., 2007). They are considered good indicators of the overall ecosystem health (Water Framework Directive, 2002) because of their close association with the stream bed, their relatively long life cycles, limited mobility (Pan et al., 2012) and because they are sensitive to environmental changes. Macroinvertebrates reflect stream conditions, integrate human and natural stressors over a long period of time and thus give a good representation of the quality of their surroundings (Cairns and Pratt, 1993). Their presence, abundance and activities are a representation of the water quality and may effectively reveal the ecological status of the ecosystem (Bredenhand and Samways, 2008).

The effects of hydro-morphological and physico-chemical conditions on macroinvertebrate communities at different spatial scales have been well-studied. For example, some studies assessed the responses of macroinvertebrates to nutrient levels (e.g. total phosphorus (TP) and nitrate (NO_3^-) (Smith et al., 2007), salinity (Eggermont et al., 2006) and sediment (Brown et al., 2000). Next to chemical variables, some studied the relationship between macroinvertebrate and habitat characteristics (Tolonen et al., 2001) as well as land cover (Black et al., 2004; Utz et al., 2009; Cuffney et al., 2010). The understanding of macroinvertebrate communities in relation to their environment is of great significance to water quality managers to identify the most appropriate actions for a successful river restoration or protection of non-impacted sites (Theodoropoulos et al., 2015).

Portoviejo is considered as the center of economic, political and cultural events in the province of Manabi. During recent years, a high pressure has been exerted on water quality and natural ecosystems in the Portoviejo River basin driven by the growing population and increasing anthropogenic activities. The pollution of the Portoviejo River causes scarcity of clean water for domestic consumption and irrigation, loss of fishing grounds (Párraga and Aguirre, 2010) and strongly affects biodiversity and mangrove ecosystems in the Portoviejo River Estuary (ACBIO, 2012). There are only a few studies that assessed the water quality based on macroinvertebrate communities in Ecuadorian watersheds. However, the Portoviejo River is an understudied area. To our knowledge no research has been carried out on the assessment of water quality and macroinvertebrate communities in this watershed. Due to the high anthropogenic impacts that have been reported, there is an urgent need to monitor and assess the water quality and biodiversity of the Portoviejo River, as this information is needed for planning and management. Therefore, the main objective of this study was to assess the ecological water quality of the Portoviejo River based on the benthic macroinvertebrate communities. More specifically, we attempted to analyze the change of community structure along key environmental variables and to find potential indicator taxa of macroinvertebrate communities in the Portoviejo River. This information could be used to establish priorities for conservation efforts for the Portoviejo River and other similar river basins, where water resources are suffering from multiple threats.

6.2 Data analysis

All statistical analyses were done using R software (version 3.2.3) (R Core Team, 2015). The protocol for data exploration as described by Zuur et al. (2010) was used to avoid common statistical problems. Prior to the actual data analysis, the initial data set was tested for outliers and correlations between explanatory variables. Following the procedure suggested by Zuur et al. (2010), one sampling location which is located close to the river mouth with five extreme high and low values compared to the majority of observations (e.g. conductivity = 49384 µS/cm, TN =below detection limit, elevation = - 4 m a.s.l.), was discarded from the analysis. To obtain a more complete understanding of the community structure, species abundances were calculated, and taxonomic richness and Shannon–Wiener Diversity Index (Shannon and Wiener, 1949) were computed using the Vegan package (Oksanen et al., 2016) for each sampling site.

Spearman's correlation coefficient is a non-parametric technique, which was used to explore the relationship among physico-chemical variables and the water quality index (Appdendix 6.1). Scatter plots were made to visualize the relationship between the BMWP-Colombia water quality index and all measured variables. A Mann–Whitney U-test was used to compare physico-chemical variables between more impacted and less impacted sampling sites. A Kruskal–Wallis test followed by post hoc multiple comparisons was performed to test whether significant differences in ecological indices existed between different types of land use, dominant substrate, sludge layer and pool-riffle pattern.

Threshold Indicator Taxa ANalysis (TITAN) was used to detect community responses to the environmental gradients in the Portoviejo River.

6.3 Results

6.3.1 Physico-chemical water quality

Table 6.1 and Figure 6.1 shows the water quality results measured at sampling sites within the Portoviejo River. High oxygen levels were observed for most sampling sites. The lowest water velocity (0 m/s) was measured at the reservoir. There was a significant, negative correlation between elevation and conductivity, NO_3^-, NO_2^-, oPO_4^{3-}, TP and TOC (r = -0.84, r = -0.81, r = -0.78, r = -0.76, r = -0.73, r = -0.86, respectively) (Appendix 6.1). The lower conductivity values (< 600 µS/cm) were observed at upstream sites, while higher conductivity values (900-2447 µS/cm) were measured at the impacted and more downstream sites. Based on the Mann–Whitney

U-test (p < 0.05), it was found that conductivity and concentrations of NO_3^-, NO_2^-, oPO_4^{3-}, TN, TP, TOC and chlorophyll a were significantly higher for more impacted sites compared to reference (less impacted) sites.

Table 6.1 Mean, median, maximum, minimum values and standard deviation of continuous environmental variables measured in the Portoviejo River and their Spearman's Rank correlation coefficients with the BMWP-Colombia index (* p<0.05, ** p <0.01).

Variable	Mean	Median	Max.	Min.	Std	*r*
Velocity (m/s)	0.4	0.4	0.9	0.0	0.3	0.48^{**}
Temperature (°C)	28	28	31	26	1.4	-0.38^{*}
Conductivity (µS/cm)	880	385	2447	164	722	-0.39^{*}
pH	8	8	9	7	0.4	- 0.05
Dissolved Oxygen (mg/L)	8	8	18	2	2.5	0.08
Chlorophyll a (mg/L)	13.5	7.2	55.2	1.9	15	-0.47^{**}
Turbidity (NTU)	15	12	34	0	11	0.09
BOD_5 (mg/L)	3.0	2.9	5.9	0.8	1.5	- 0.21
Nitrate-Nitrogen (mg/L)	1.1	0.5	2.8	0.2	0.9	- 0.27
Nitrite-Nitrogen (mg/L)	0.1	0.1	0.1	0.0	0.1	- 0.33
Ammonium-Nitrogen (mg/L)	0.1	0.1	0.2	0.04	0.04	0.30
Total Nitrogen (mg/L)	1.8	1.2	5.7	0.5	1.5	- 0.18
Orthophosphate (mg/L)	0.2	0.2	0.3	0.1	0.1	- 0.17
Total Phosphorus (mg/L)	0.2	0.2	0.5	0.1	0.1	-0.39^{*}
Total Organic Carbon (mg/L)	15.8	16.8	37.7	3.0	10.4	- 0.35
Elevation (m.a.s.l.)	61	59	121	0	37	0.39^{*}

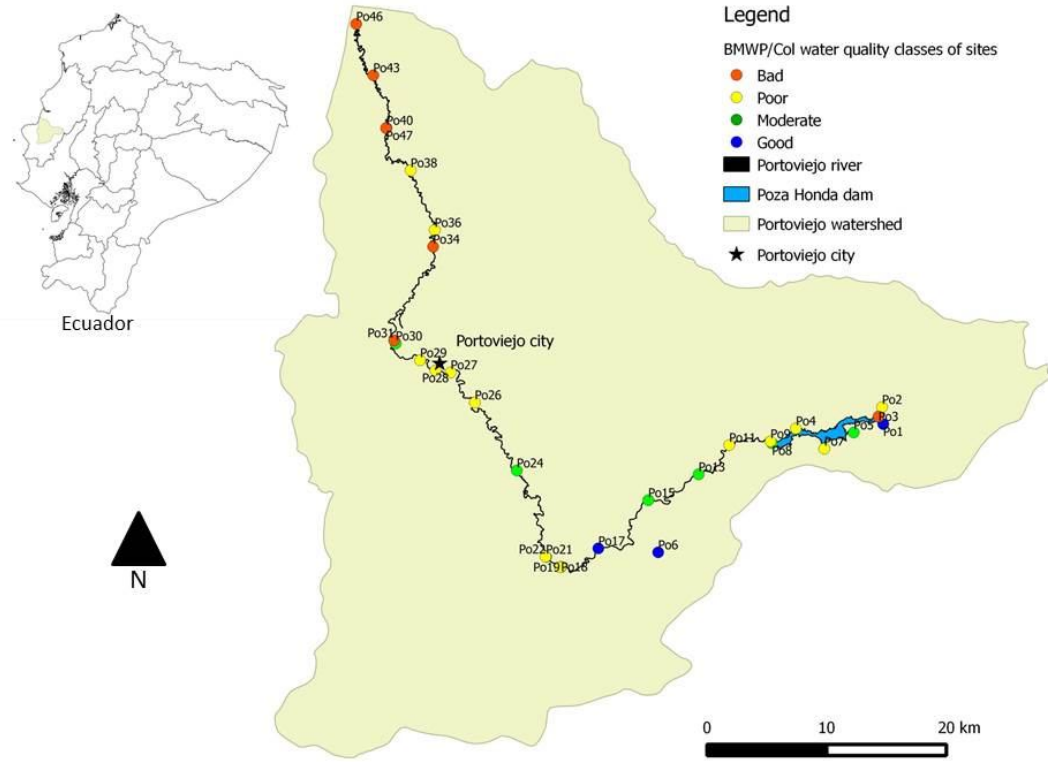

Figure 6.1 Map of the study area of the Portoviejo River with indication of the ecological water quality based on the BMWP-Colombia for each sampling site

Lower BMWP-Colombia score was observed for sites characterized by high conductivity, high chlorophyll a, high nitrate-nitrogen and high nitrite-nitrogen (Figure 6.2).

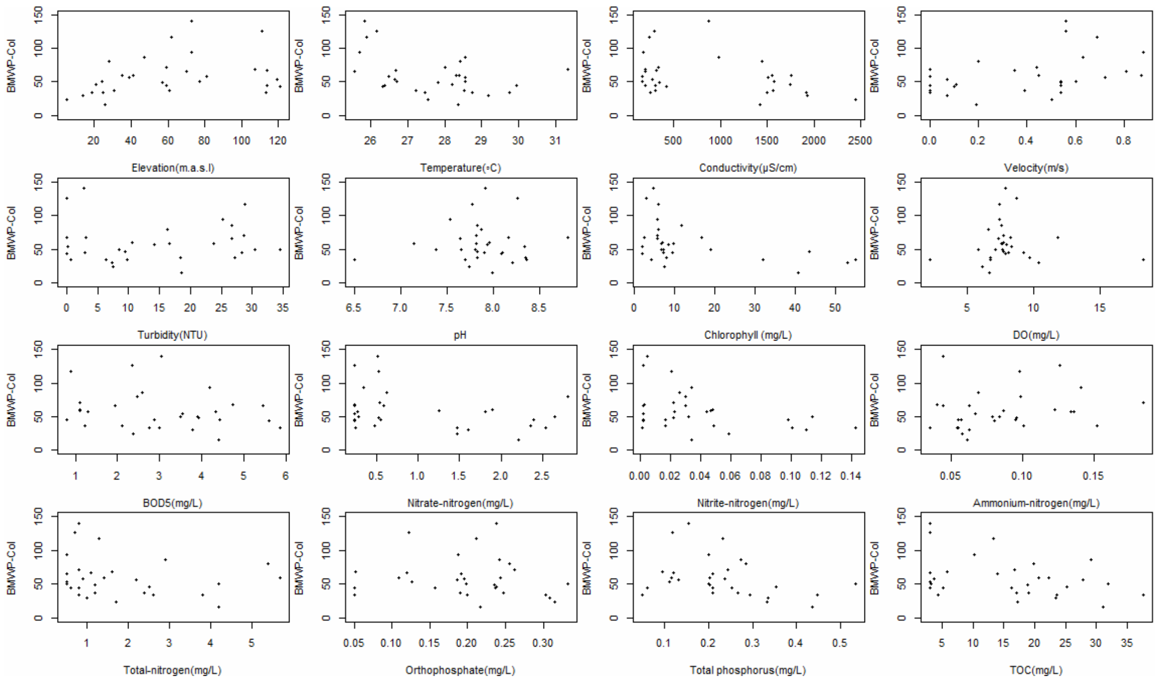

Figure 6.2. Plots of physical–chemical variables in relation to BMWP-Colombia for sampling sites in the Portoviejo River.

6.3.2 Relationship between macroinvertebrate communities, physico-chemical conditions and habitat characteristics

In total, more than 8,300 individuals belonging to 53 macroinvertebrate families were found (Appendix 6.2). The taxon richness is varying from 4 to 22 taxa per sampling site. Chironomidae was the most frequently encountered taxon, followed by Coenagrionidae and Libellulidae (29, 21 and 20 sites, respectively). Thiaridae was the most abundant taxon, followed by Chironomidae (5231 and 805 individuals, respectively). Based on the BMWP-Colombia scores, the sampling sites of the Portoviejo River were categorized into four water quality classes: good, moderate, poor and bad (Fig.6.1). The Shannon-Wiener index ranged from 0.23 to 2.58. There was a strong positive correlation between the BMWP-Colombia scores and taxonomic richness (S) (Spearman's correlation coefficient = 0.94). Spearman's correlation coefficient between BMWP-Colombia and Shannon’s diversity index (H) was 0.58. However, the Spearman's correlation coefficient between the BMWP-Colombia and taxa abundance was only 0.30. The highest BMWP-Colombia value (140) was recorded at one sampling site where the taxonomic richness was also the highest (22 taxa). This location is surrounded by forest, has gravel substrate and a moderately developed pool-riffle pattern.

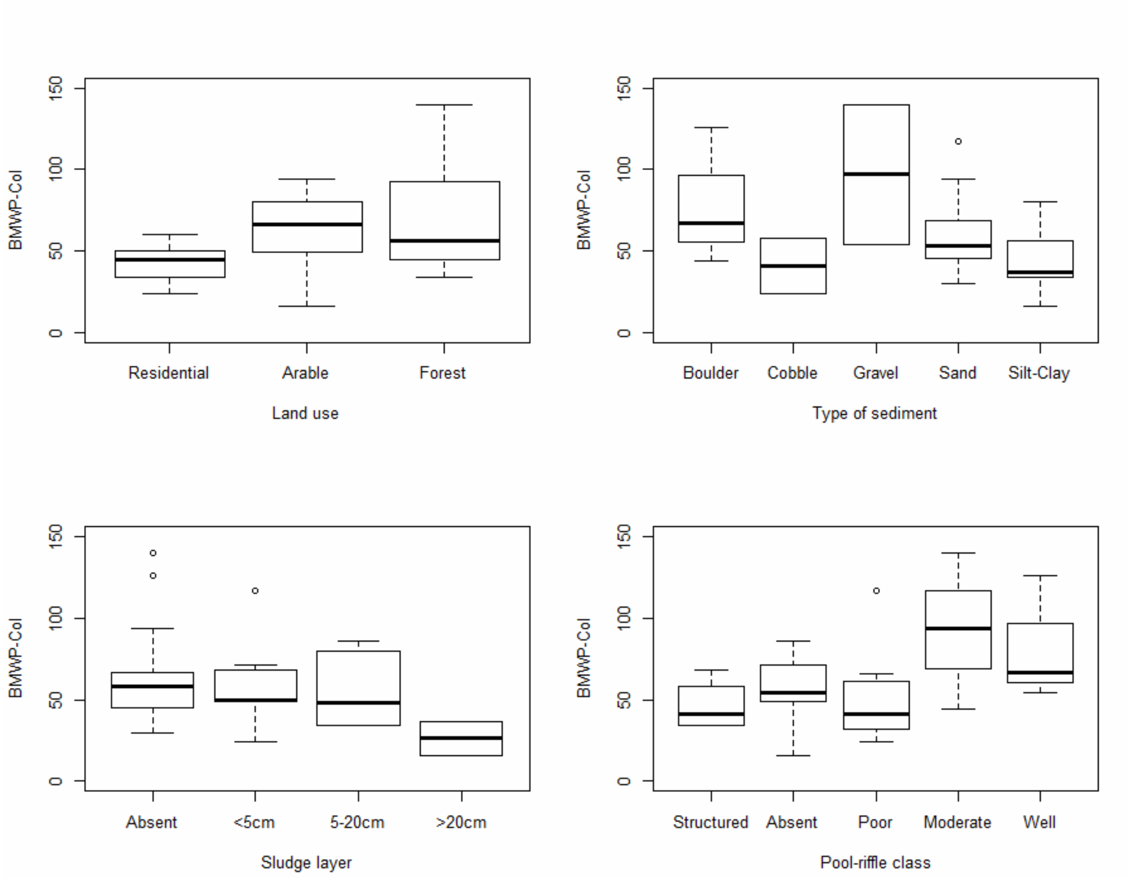

Figure 6.3. Boxplots of the different classes of land use, type of sediment, sludge layer and pool–riffle class in relation to BMWP-Colombia for sampling sites in the Portoviejo River. Bold horizontal lines represent median, boxes represent interquartile ranges (25–75% percentiles) and range bars show maximum and minimum values, small black dots show outliers.

The Spearman's rank correlation coefficients between the biological water quality index (BMWP-Colombia) and the physico-chemical variables indicated that the BMWP-Colombia scores were positively correlated with elevation and stream velocity. In addition, BMWP-Colombia values showed a negative association with temperature, conductivity, chlorophyll a, NO_3^-, NO_2^-, TP and TN (Table 6.1). Higher BMWP-Colombia values were observed at sampling sites surrounded by arable land use, gravel sediment, absence of a sludge layer and at least a moderately developed pool-riffle pattern. However, statistical analysis did not reveal any statistically significant differences.

6.3.3 Threshold change points and indicator taxa

TITAN was used to evaluate the variation in taxonomic composition of macroinvertebrate communities in response to all physico-chemical variables. However, due to the low number of reliable indicator taxa, TITAN could only reveal the community change along the gradient of elevation, conductivity and nitrate-nitrogen concentration (Fig. 6.4). Change point analysis identified a negative response of macroinvertebrate communities (sumz-) at an elevation below 30 m a.s.l. (Fig. 6.4A), conductivity values above 1200 µS/cm (Fig. 6.4C) and nitrate-nitrogen values higher than 0.6 (mg/L) (Fig. 6.4E). In addition, the change points revealed a positive response of macroinvertebrate taxa (sumz+) at elevations above 62 m a.s.l. (Fig. 6.4A), conductivity values below 1433 µS/cm (Fig. 6.4C) and nitrate-nitrogen values lower than 2.3 (mg/L) (Fig. 6.4E). Atyidae, Thiaridae and Corbiculidae displayed a negative response to an increasing elevation, while they showed a positive response to the increase of conductivity and nitrate-nitrogen. In contrast to those taxa, Acari, Baetidae and Leptohyphidae showed a positive response to an increasing elevation (Fig. 6.4B) and displayed a negative response to an increasing conductivity (Fig. 6.4D) and increasing nitrate-nitrogen values (Fig. 6.4F). Naucoridae showed a positive response to an increase in elevation (Fig. 6.4B), whilst Veliidae and Libellulidae showed a negative response to an increase of conductivity and nitrate-nitrogen, respectively (Fig. 6.4D and 6.4F).

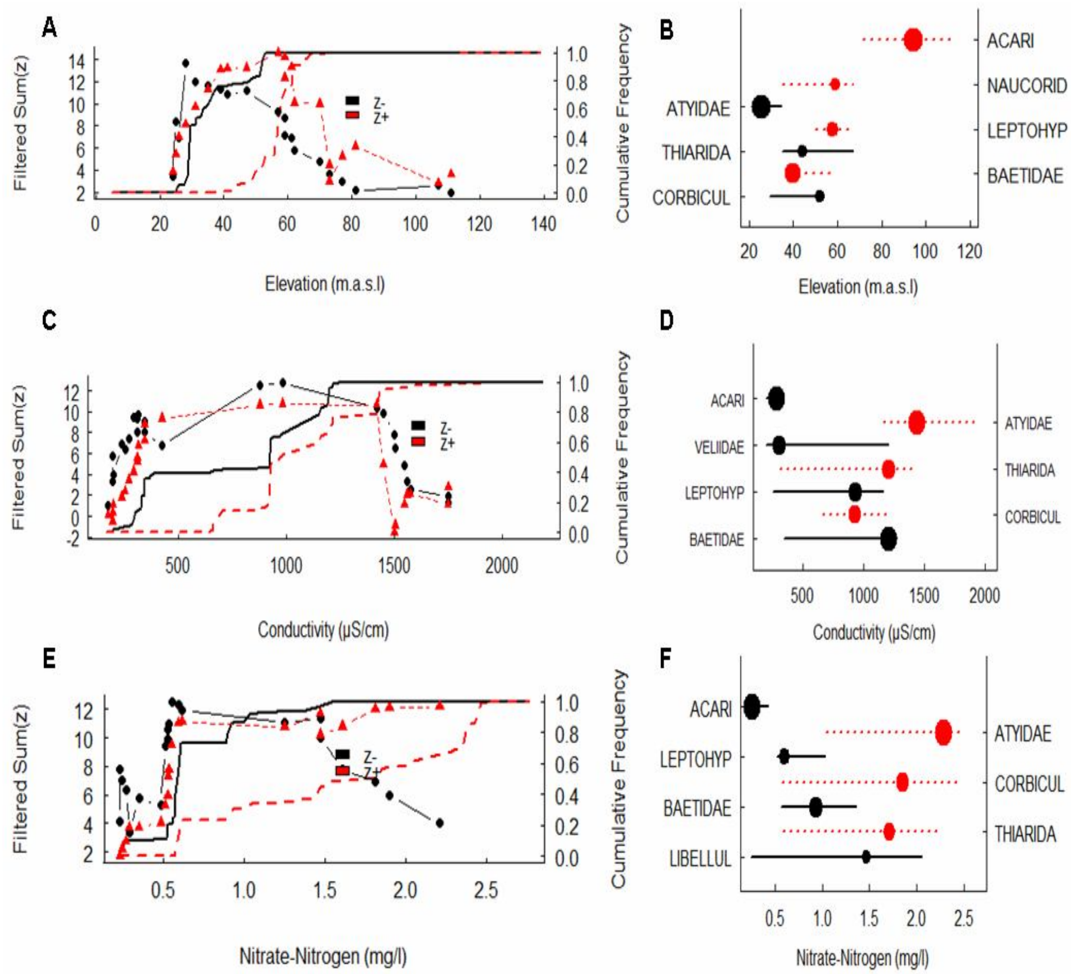

Figure 6.4 Threshold Indicator Taxa Analysis and change points (dots) for macroinvertebrate community response to elevation (A,B), conductivity (C,D) and nitrate-nitrogen (E,F) gradient ($p \leq 0.05$, purity = 0.95, reliability = 0.95 for 5 minimum number of observations, 1000 bootstrap and 1000 permutation replicates). Negatively associated taxa (z–) are indicated by black symbols and lines, and positively associated taxa (z+) are indicated by red symbols and dashed lines. Solid and dashed lines are cumulative frequency distributions of sum(z-) and sum(z+) maxima (respectively) across bootstrap replicates. Size of change point symbol (dots) is proportional to the magnitude of the taxa response. Z− species responded negatively to an increase in the environmental gradient, Z+ species responded positively to an increase in the gradient. Horizontal lines suggest 5-95% quantiles from the bootstrapped change point distribution.

ACARI = Acari, ATYIDAE = Atyidae, BAETIDAE = Baetidae, CORBICUL = Corbiculidae, LEPTOHYP = Leptohyphidae, LIBELLUL = Libellulidae, NAUCORID = Naucoridae, THIARIDA = Thiaridae, VELIIDAE= Veliidae.

6.4 Discussion

6.4.1 Water Quality of the Portoviejo River

The majority of the sampling locations have DO concentrations ranging from 7 to 10 mg/l. The lowest value of DO (2.2 mg/L) was observed at the Poza Honda reservoir, which was characterized by the highest BOD_5 (5.9 mg/L), a sludge layer between 5-20 cm, a water hyacinth coverage of 50-75% and stagnant water. Based on the ecological water quality index (BMWP-Colombia), this sampling site had a bad water quality. This finding may be explained by a combination of unfavorable conditions and the fact that when the percentage of water hyacinth cover is higher than 50%, the cover might be too dense, which negatively affects the physico-chemical water quality (e.g. DO, BOD_5) (Nguyen et al., 2015). Damming causes unfavorable changes in the riverine biota through changes in flow regime, sediment transport and habitat modification (Käiro et al., 2011). As the Poza Honda dam started operating in 1971, there might be sediment accumulation, which can negatively affect the macroinvertebrate community. Moreover, the interconnection between dams caused the wide spread of the invasive water hyacinth from the Daule-Peripa Reservoir (Guayas River) where about one third of the Daule-Peripa reservoir is invaded by water hyacinth (*Eichhornia crasssipes*) (Gerebizza, 2009) to the Poza Honda Reservoir. Unfortunately, detailed information about the impact of water hyacinth on water quality in the Poza Honda Reservoir and Portoviejo River are not available.

The negative correlation between elevation and conductivity, nutrient concentrations (e.g. NO_3^-, NO_2^-, oPO_4^{3-}, TP) and TOC indicates the cumulative negative impacts of human disturbance on water quality from upstream to downstream in the Portoviejo River. The Portoviejo River suffers from a high level of environmental deterioration caused by deforestation, burning of vegetation, drainage of agrochemicals and fertilizers, garbage disposal and discharge of sewage without adequate previous treatment (Párraga and Aguirre, 2010; ACBIO, 2012). As such, nutrients (e.g. NO_3^-) can get into the water as a result of domestic wastewater discharges, agricultural activities (e.g. using manure and fertilizer containing NO_3^-) and from oxidation of nitrogenous waste products in human and animal excreta

(Singh and Sharma, 2014). Moreover, Wang et al. (2012) emphasized that watershed urbanization levels are significantly associated with increased conductivity, total nitrogen, ammonia, phosphate and chemical oxygen demand. Our observations are in agreement with Dominguez-Granda et al. (2011a) that low conductivity values were measured at sites with low human impact in the Chaguana River basin, Ecuador. The increase of human activities negatively affects the water quality in the Portoviejo River which is shown by the increase of nutrient concentrations (NO_3^-, NO_2^-, TN and TP) and conductivity in the downstream locations.

We found that the lower BMWP-Colombia scores were reported for sites located near residential areas and right after the outlet of a new municipal wastewater treatment plant of the Portoviejo city; chlorophyll a value was higher than 30 mg/l at those sites. High concentration of chlorophyll is likely to be caused by the discharge of algae from the wastewater stabilisation pond. It is estimated that each year roughly 20 million cubic meters of wastewater are discharged into the Portoviejo River (ACBIO, 2012). As such, the Portoviejo River continues to suffer from sewage pollution. Indeed, the increase of nutrient levels by domestic waste of residential areas with incompletely treated wastewater probably caused poor water quality by an increase of phytoplankton abundance (expressed as chlorophyll a concentration). Sampling sites at the higher reach of the river are less affected by anthropogenic disturbances; hence more diverse macroinvertebrate communities are present in that area due to a better chemical water quality status. Our results are also in line with Jacobsen (1998), who found that the BMWP-Colombia index tended to increase at the upstream sites and decrease at the downstream sites in the dry season in highland streams in Ecuador. The results of the ecological indices are in accordance with our expectations and other research results. Our study, in accordance with others, indicates strong negative effects of anthropogenic disturbances on biological diversity and water quality in the Portoviejo River.

6.4. 2 Relevance of the TITAN method: Indicator taxa of environmental change and threshold for macroinvertebrates

In the present research, we show for the first time the thresholds for the changes in macroinvertebrate communities along an environmental gradient in the Portoviejo River. TITAN uncovered a clear community change along the gradient of elevation, conductivity and nitrate-nitrogen. Furthermore, we revealed a community threshold for salt-sensitive taxa at 1200 µS/cm and salt-tolerant taxa at 1433 µS/cm, which is close to the findings of Schroder et al. (2015), who determined a community threshold for salt-sensitive taxa at 926 µS/cm and community threshold for salt-tolerant taxa at 1464 µS/cm in the Lippe River, Germany. Following the German chemical classification of conductivity (Schroder et al., 2015), in the range from 725 – 1207 µS/cm, water becomes critically saline and thus less suitable for certain macroinvertebrates. This result is in line with the findings of Dunlop et al. (2005), who investigated that the shift of macroinvertebrate communities with a high proportion of salinity-sensitive taxa to communities of more tolerant individuals is between 800 and 1000 µS/cm in Queensland, Australia. Indeed, conductivity is an important variable which affects macroinvertebrate community composition (Boets et al., 2010). Moreover, conductivity was a key variable to the shift of macroinvertebrate, according to the decision trees applied in Flanders (D'heygere et al., 2003). In the present study, there was a strong relationship between conductivity and NO_3^- (r = 0.82), NO_2^- (r= 0.75) and TOC (r = 0.73). It is possible that the increase of nutrient levels are a factor which promoted individual taxon (and thus community) conductivity thresholds (Schroder et al., 2015). Our results, in agreement with other studies, imply that the conductivity thresholds identified could be applied in similar river systems. Community thresholds may be applied to define conductivity thresholds in order to protect the macroinvertebrate community in the Portoviejo River. As such, this information has a valuable implication because the degradation of river water quality in Ecuador could be addressed more effectively.

In this study, TITAN identified the points at which the macroinvertebrate community negatively responds to an elevation of 30 m a.s.l., a nitrate-nitrogen value of 0.6 mg/L and the points corresponding to a positive effect at elevations of 62 m a.s.l. and a nitrate-nitrogen value of 2.3 (mg/L). However, TITAN did not find strong evidence of a threshold for stream velocity, pH, TN, TP and organic matter (DO,

BOD_5 and TOC), as biological communities are always affected by multiple factors and it is not possible to separate the effects of each factor when they co-vary (Berger et al., 2016). The difference between community thresholds and biodiversity trends could be explained by the fact that aggregate metrics (e.g. biodiversity) are confounded by synchronous changes in tolerant and sensitive taxa (King et al. 2011, Baker and King 2013). Therefore, it is not possible to consider a single threshold value to explain the change of the entire community. Differences of elevation and habitat conditions lead to differences in macroinvertebrate assemblages among river systems. In this study, the elevation ranged between 0 and 121 m.a.s.l., which is very low. In addition, there is a negative correlation between elevation and conductivity, nutrient concentrations and TOC. In the literature, it has been shown that there is a strong relation between nutrient concentrations and macroinvertebrate communities (Ashton et al., 2014). Moreover, community composition was related to conductivity and mean velocity (Black et al., 2004). As such the reported tipping points for elevation may be not representing the influence of altitude on the community, but indicating the cumulative negative impacts of human disturbance on water quality from upstream to downstream in the Portoviejo River. For sustainable river management, it is recommended to limit the nutrient and chemical loads into the river, but also to protect the natural hydromorphological conditions.

TITAN revealed that Atyidae, Corbiculidae and Thiaridae showed a positive response to the increase of conductivity and nitrate-nitrogen, whilst they were assigned a negative response to an increasing elevation. In contrast to those taxa, Acari, Baetidae and Leptohyphidae showed exactly opposite responses. Based on the present analyses, the three tolerant taxa are widely distributed and are considered invasive alien species in several parts of the world. Atyidae is a family of shrimp in the order Decapoda, which occur in all tropical and most temperate waters of the world (Torati et al., 2011). Corbiculidae is a family of clams in the order Veneroidea (Gofas, 2015), that were introduced to America from Asia (Strayer, 1999). Thiaridae is a family of snails in the order Gastropoda, which is pantropically distributed including Central and South America, Africa, South East Asia, Caribbean Islands, Pacific and Australia (Glaubrecht et al., 2009).

The sensitive taxa, Leptohyphidae and Baetidae mayflies, belong to the order Ephemeroptera. The mayfly family Leptohyphidae is indigenous in South American and widely distributed throughout North, Central, and South America (Baumgardner and Ávila, 2006). Baetidae have a global distribution and a high species diversity (Múrria et al., 2014). In general, Ephemeroptera are typically classified as a sensitive group. Our finding is supported by a previous study, which indicated Ephemeroptera as a salinity sensitive indicator (Schroder et al., 2015). Veliidae belonged to the order Hemiptera and live in a wide variety of aquatic habitats. Veliidae comprise a globally distributed family of predatory semi-aquatic bugs with more than 960 known species and the Neotropical fauna corresponds to approximately 30% of the total species (Moreira and Barbosa, 2011). The dragonfly family Libellulidae, is also broadly distributed throughout the world (Heckman, 2008). The family of Naucoridae (Creeping Water Bugs) is found in stagnant and running freshwater in tropical regions, with the highest diversity in the Neotropics (Zettel and Lane, 2011). All the indicator species identified in this study have a worldwide distribution; therefore, they could be used as potential indicator taxa of different ecoregions. TITAN has the potential to inform decision makers about critical levels of anthropogenic changes, which correlated with great changes in ecological communities (King and Baker, 2014). Each sensitive taxon displayed a significant increase or decrease at a specific threshold. As such, the TITAN results show that many taxa could increase or decrease with only a small change in certain environmental variables. For example, in this study, the dragonfly family Libellulidae has a nitrate-nitrogen threshold value of 1.5 mg/L. The adults and larvae of dragonflies are able to help to control endemic diseases (e.g. malaria, yellow fever and dengue fever) (Heckman, 2008) as they are important predators of for example mosquito larvae. As such, it is important to manage the nitrate-nitrogen level in order to control the presence of dragonflies and consequently also the presence of diseases.

TITAN uncovered a clear community change along both human affected gradients (e.g. conductivity and nitrate-nitrogen) and natural gradients (e.g. elevation). The negative taxa for the increase of pollution (e.g. conductivity and nitrate-nitrogen) are the positive taxa for the increase of elevation. The indicator taxa revealed by thte method are in line with several related literature sources and general ecological insights on these taxa. TITAN provided thus additional evidence for the community shifts due to the change of water quality along the Portoviejo River. Among the tolerant taxa, three widely distributed taxa were present: Atyidae, Thiaridae and Corbiculidae. Moreover, these taxa are also considered to be invasive, which highlights the importance of identifying tipping-points in order to conserve sensitive species (Schroder et al. 2015). With increasing conductivity and nitrate-nitrogen, sensitive taxa like Baetidae and Leptohyphidae will show a decrease in abundance and frequency of occurrence, which will reduce the indigenous community composition and allow for invasive species to take over. Based on these observations, management related to aquatic conservation, biological invasions, ecosystem restoration and natural resources can be performed (King & Baker, 2010). Moreover, these tipping-point values have valuable applications for detecting reference condition boundaries such as the conductivity and nitrate nitrogen should not exceed the threshold values and selecting sites at greatest risk of significant change (e.g. conductivity higher than 1400 µS/cm and nitrate nitrogen value is higher than 2.3 mg/L). Nevertheless, the obtained tipping-point values should be considered as primary results as this is only the first work on macroinvertebrate communities in the Portoviejo River.

In this study, elevation, conductivity and nitrate-nitrogen concentrations appeared to be strongly linked to the macroinvertebrate community composition. To our knowledge, water quality issues, aquatic ecosystems and ecosystem services received very limited attention in Ecuador. No standardized sampling procedures and no environmental monitoring program are available to assess the water quality (Nolivos et al., 2015) resulting in limited availability of information on the physical, chemical and ecological status of Ecuadorian rivers (Andres, 2009; Nolivos et al., 2015). Therefore, the thresholds relevant to the macroinvertebrate communities' response in the Portoviejo River will be particularly useful for early warning in Ecuadorian rivers. However, this is the first publication on macroinvertebrate communities in the Portoviejo River and this region. The results may serve as a starting point for future study, which is needed to check if the thresholds and indicator taxa identified from this case study are consistent and applicable at a larger scale.

6.5 Conclusions

In the present chapter, we provided baseline information about the physico-chemical water quality, the hydromorphological conditions and the macroinvertebrate community composition in the Portoviejo River (Ecuador). The BMWP-Colombia scores showed that water quality of the sampling sites within the Portoviejo River ranged from good to bad. TITAN was used to relate macroinvertebrate community composition changes with the physical-chemical and hydromorphological conditions in the river. We revealed clear tipping points in elevation, conductivity and nitrate-nitrogen concentrations and associated indicator taxa. Atyidae, Corbiculidae and Thiaridae showed a positive response to the increase of conductivity and nitrate-nitrogen, whilst they were assigned a negative response to an increasing elevation. In contrast to those taxa, Acari, Baetidae and Leptohyphidae showed a negative response to the increase of conductivity and nitrate-nitrogen, whilst they were assigned a positive response to an increasing elevation. Based on these correlations, these taxa can be considered as indicator taxa to assist in identifying environmental thresholds of the Portoviejo River. TITAN could provide a useful management tool to support water quality management in the status of the Portoviejo River and similar rivers in the region. Indeed, based on the patterns that are characterized in this research novel management approaches can be developed and implemented.

Chapter 7: General discussion and further research

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Abstract

The overall results from this PhD study demonstrate specific practical issues associated with the impacts of hydropower generation on river ecosystems and water quality management. The main issues are the intensive development of water hyacinth on the hydropower reservoirs and the accumulation of multiple human impacts affecting ecological water quality and the composition of the associated macroinvertebrate communities. This chapter provides a general discussion about the research methodologies used in data collection, application of ecological modeling, ecological assessment based on macroinvertebrates in hydropower dam impact assessment and practical issues in river management. It also suggests recommendations for further research about hydropower dam impacts.

7.1 Introduction

The results from this PhD study demonstrate several practical issues associated with the impacts of hydropower generation on river ecosystems and water quality management in Ecuador. Hydropower generation causes river fragmentation in its creation and development of a hydropower reservoir. The case study of the Guayas river basin showed that certain physical-chemical variables (e.g. stream velocity, chlorophyll a, conductivity, temperature and elevation) were the major environmental factors influencing the distribution of macroinvertebrate populations and species specific habitats (c.f. chapter 5). On a local scale (with a focus on Daule-Peripa reservoir), habitat characteristics (the presence of water hyacinth) was an important variable affecting the diversity of macroinvertebrates of the reservoir (c.f. chapter 4). However, in the Daule-Peripa reservoir, the cause-effect relationships between the presence of water hyacinth and environmental variables remain unclear. The threshold indicator taxa analysis showed the shift between the macroinvertebrate communities related to the change of stream velocity (c.f. chapter 5) and elevation, conductivity and also nitrate nitrogen concentrations (c.f. chapter 6). In the Portoviejo river basin, the BMWP-Colombia scores were positively correlated with elevation and stream velocity while they showed a negative correlation with temperature, conductivity, chlorophyll a, NO_3^-, NO_2^-, TP and TN. However, TITAN revealed clear tipping points in elevation, conductivity and nitrate-nitrogen concentrations. Results of these individual analyses were presented and discussed in detail in each chapter. Our results indicated the importance of multiple factor assessments at various spatial scales for better understanding the impacts of hydropower generation on river water quality and biodiversity.

Hydropower dams can produce huge environmental impacts induced by the change in river morphology, physical-chemical conditions and biological composition and ecosystem functioning. However, in this study, many effects of hydropower dams on rivers such as sedimentation, thermal stratification and eutrophication in the reservoir have not yet been assessed. In addition, the ecological impacts of hydropower dams on other resident communities such as algae, amphibians, fishes and birds have not been included in this study. Nevertheless, assessments of the shifts in plant cover and macroinvertebrate communities are major steps towards a more integrated biological assessment and a wider, more balanced understanding.

The aim of this chapter is to link the results and discussions in the previous chapters and present some general aspects regarding the application of ecological models for dam impact assessment and river management. This chapter will discuss some practical issues, including data collection, model development, assessment methods and recommendations for river management. The dissertation ends with some recommendations for further research on this topic.

7.2 Data collection

Data collection is one of the crucial steps in impact assessment. However, many statistic predictive modeling exercises are still based on field data from observational studies and still lacking a designed sampling strategy (Guisan and Zimmermann, 2000). Here, we will discuss some useful considerations for ecological impact assessment of hydropower generation related to data collection.

Firstly, we would like to address where the samples should be collected. Dams are known for their impacts on ecosystems at the catchment scale, with both upstream and downstream effects stemming from inundation, flow manipulation and fragmentation (Nilsson et al., 2005). The results from previous chapters of this study show the cumulative negative impacts of human disturbances on water quality and biological diversity from upstream to downstream of a river. Moreover, some sampling sites within the associated tributaries were indicating severe water quality problems, for example, low water levels and high conductivity values. Therefore, the sampling sites should be taken along the river, including: river at upstream reservoir, hydropower reservoir and river below the reservoir at both the main river and its main tributaries. A case study from Schinegger et al (2012) showed that various human pressures affected running water ecosystems in Europe. In total 47% of the sites are multi-impacted and 90% of lowland rivers are impacted by a combination of all four pressure groups (hydrology, morphology, water quality and connectivity) (Schinegger et al., 2012). Results of this study also showed that hydropower dams are one of the main drivers causing detrimental changes in the riverine biota through changes in the flow regime. At the same time, Ecuadorian rivers are facing many other problems such as changes in land use and intensification of agriculture. Hence, target sampling sites should represent a range of different impacts and various levels of water quality from polluted to unpolluted sites. The distribution of macroinvertebrates constitutes a complex process associated with many factors at different scales. Macroinvertebrate community analysis revealed river degradation due to different anthropogenic disturbances and habitat destruction. In addition, the small number of samples in the Portoviejo river basin (31 sites) limited the ability to transfer the results to other watersheds. Hence, it is useful to assess the ecological impact at the river and river basin scales in order to obtain a better understanding of the relationship between macroinvertebrate communities and environmental conditions. In other words, a significant number of samples is needed if a diverse set of human activities is putting pressure on the river system. In general, most studies were based on a very limited number of sampling sites. They often only focused on the assessment of an area near to the dam. Consequently, results reported tend to underestimate or overestimate the impacts of dams on upstream and downstream sites.

Secondly, we would like to focus on the variables that should be measured because this is crucial for model development. The appropriate selection of sampling variables is crucial because different models can be developed for the same taxon when different databases are used (Goethals, 2005). Furthermore, the type of variables collected clearly influenced the derived data driven models (Goethals, 2005). Our results pointed out the importance of carrying out multiple factor assessments for reliable predictions of habitat suitability. Results from chapter 4 indicated that water hyacinth was present at sites with a low turbidity of what ??. Surprisingly, there is no clear relationship between the presence of water hyacinth and temperature, conductivity, TDS, pH, chlorophyll a, chlorides, DO, oxygen saturation. Note that a major limitation of this study was the lack of data on water column nutrients for the Guayas river basin. Therefore, the cause-effect relationships between the presences of water hyacinth and environmental variables remain unclear. Nevertheless, the presence of water hyacinth was a major variable affecting the diversity of macroinvertebrates in the Daule-Peripa hydropower reservoir. Chapter 5 indicated that chlorophyll a, conductivity and stream velocity were key drivers affecting the change in the macroinvertebrate community while chapter 6 suggested that conductivity and nitrate-nitrogen concentrations were influential variables, which needed to be considered as important ecological drivers. Paulsen et al. (2008) revealed the most widespread stressors for river in the United States were known to be nutrients (e.g. nitrogen and phosphorus), riparian disturbance and streambed sediments. Ecuadorian rivers in the past and up to the present are facing many threats such as gravel mining, gold mining, deforestation, hydroelectric projects and contamination (Ecuadorian Rivers Institute, 2016). Although flow is the key driver in hydropower dam impact, other factors such as conductivity and nutrients are also major concerns in structuring macroinvertebrate communities. In-depth analyses of relationships among various pressure types such as hydrology, continuum disruption and combined pressure effects, and linkages to biotic classifications may also yield a better understanding of restoration and mitigation requirements (Paulsen et al., 2008). Therefore, it is recommended to measure hydro-morphological, physical-chemical variables, biological and habitat characteristics a synthesis assessment should be implemented in order to understand the multiple spatial, temporal and interactive impacts of dams. Several types of variables are needed for integrated impact assessment. However, this can be expensive if multiple variables have to be measured, especially when the research is covering a large and complicated topography area (e.g. Guayas River basin). Therefore, from a practical point of view, it is needed to find the key drivers and work further on the economically appropriate and easy collectable variables such as velocity and conductivity, depth and DO.

Thirdly, we discuss when samples should be taken. In this study, both of the sampling campaigns in the Guayas and Portoviejo river basins were performed only one time in the dry season for safety reasons and also accessibility to the sampling sites. It was estimated that the highest conductivity value, the lowest water level and other extreme values were obtained during the dry season. It was assumed that the worst conditions occur in the dry season, causing harsh conditions for macroinvertebrates and other communities in the ecosystem. Under those particular circumstances, the results were adequate for water quality assessment purposes. Since both the Daule-Peripa and the Poza Honda dams were designed for multiple purposes, including hydropower and irrigation, the discharge might be managed in different ways. For irrigation purpose, the discharges tend to fluctuate by season; discharges increase during the irrigation season to meet irrigation water demands, but decrease in other months to restore reservoir storage for the next irrigation season. On the other hand, hydropower dams are operated based on stated energy demands. Therefore, the water uses or seasonal management should be considered. It is advised to repeat sampling during the wet season to test this assumption made, and to check if the results are similar. Although our results from the dry season provided valuable insights, the conclusions may need to be verified.

7.3 Model development and impacts assessment

In this study, we primarily used multivariate analysis to understand which variables are important, then TITAN was applied to identify the thresholds and to elucidate how certain variables (e.g. stream velocity) influence resident macroinvertebrate communities. This approach can be applied to answer the question regarding dam and river management. For example, what are the most important factors in defining macroinvertebrate communities? Which velocities of flow are needed? Which taxa will be influenced if the stream velocity is changed? This dissertation can contribute to the transparency of the thresholds defining the process in river management. TITAN allows the identification of specific thresholds which can be applied to predict the fluctuations and changes in macroinvertebrate communities under an alternative water management scheme.

TITAN was used to detect the variation in taxonomic composition of macroinvertebrate communities in response to stream velocity in the Guayas river basin (c.f. chapter 5) and all physico-chemical variables in the Portoviejo River (c.f. chapter 6). TITAN revealed 14 and 20 taxa as the positive/negative indicators for the change of stream velocity in the Guayas river basin (c.f. chapter 5). In chapter 6, TITAN could only reveal the community change along the gradient of elevation, conductivity and nitrate-nitrogen concentration. Only three or four taxa are defined as positive/negative indicator for the change of each variable. In this case, it is possible that the thresholds and taxa responses are basin specific because of taxa presence and the ranges of investigated variable. In addition, due to the low number of reliable indicator taxa, TITAN did not find strong evidence of a threshold for temperature, pH, DO, chlorophyll, turbidity, BOD_5, TP, oPO_4^{3-}, NH_4+, NO_2, TN, TOC and stream velocity. Although, several physico-chemical variables (e.g. velocity, chlorophyll and

TP) are considered important in the distribution of macroinvertebrates, their thresholds were not discovered by the TITAN methods, most probably as a result from limited amount of sampling sites, the environmental variable range and a too limited number of cases for both good and affected sites. In addition, the results are also influenced by other factors (e.g. habitat characteristics that were not measure).

Generalized linear models were developed to investigate which variables determined the occurrence of water hyacinth. The regression modelling paradigms (e.g. generalized linear models GLM's) showed advantages in dealing with non water quality variables (e.g. land use); GLMs can be used as effective tools for the analysis of aquatic habitat-species relationships (Ahmadi-Nedushan et al., 2006). Moreover, it also provided the solution for the main bottleneck in the application of other modeling techniques, for example, the requirement of ecological expert knowledge for the implementation of fuzzy logic models (Mouton, 2008) or the need for reference conditions in the Ecological Potential assessment approach (Molozzi et al., 2012).

This study provides insights in the ecological impacts of hydropower dams on river ecosystems. The results of this work show the diverse sets of impacts on the physical habitats, chemical water quality as well as plant and invertebrate communities during hydropower operation stage. The adverse environmental impacts of hydropower generation are diverse and complex (Abbasi and Abbasi, 2000). Besides the impact of the dam as a physical structure, ie river fragmentation, the most serious detrimental impacts of hydropower dams on the aquatic ecosytems are caused by hydropeaking (Bruder et al., 2016). Hydropeaking is the intense unnatural discharge of excessive water volumes of power plants due to energy demand (Bruder et al., 2016). During periods of high energy demand, power is generated and water is rapidly released producing significantly higher and faster flows (Jones, 2014). In contrast, when energy demand is low, flow decreases and ranges from only dam 'leakage' to an imposed minimum discharge (Clarke et al., 2008). Hydropeaking causes severe daily and sub-daily fluctuations in discharge and water levels (Meile et al., 2011). As a consequence, hydropeaking quickly reduces the quality and availability of suitable habitats for aquatic organisms through dewatering, and suitable shore habitats are displaced or lost, fine sediments are re-suspended, increasing erosion and water turbidity (Tonolla et al., 2017). In this study, macroinvertebrates were used as indicators for the change of water quality. As macroinvertebrates have a limited mobility, they are more sensitive to local disturbances, and are able to detect structural changes and habitat loss, and different species have different degrees of tolerance towards pollution (Molozzi et al., 2012). Therefore, the presence or absence of specific macroinvertebrates in conjunction with ecological modeling can provide the ecological evidence to assess the impact within the context of long-term assessment and current and future river management.

Damming can create a multitude of different impacts on aquatic ecosystems. Dams often severely alter the functioning of river systems, both locally as well as upstream and downstream. The ecological impacts of the dams can begin immediately during the initial dam construction, operational phases and after dam removal. Many previous research studies have demonstrated the impact of hydropower dams on fish (Brown et al., 2013), amphibians (Yarnell et al., 2012) and phytoplankton (Silva et al., 2014). In addition, eutrophication has been extensively reported in many other tropical hydropower reservoirs such as Lake Gilgel Gibe I (Ethiopia) (Ambelu et al., 2013). However, the assessment methods used within these studes has not as yet been standardized. Marchant et al. (2006) suggested the appropriate assessment methods should: (1) provide early warning of a wide range of environmental stressors at the appropriate temporal and spatial scales, (2) indicate the cause of change and degradation status of the environment, (3) require cost-effective tools and the needed resources should be available for assessment and (4) provide easily interpretable, user friendly outputs that relate to management objectives. Moreover, a simple and accessible model is needed, because a number of the previously utilised comprehensive models are too complex for a local administrative officer in charge (Ueda et al., 2015). Under these circumstances, there is a need for the development of practical tools such as an assessment index or modeling toolbox. Therefore, we provide a checklist for impact assessment of hydropower dams. This checklist provides major contemporary research issues related to the development and application of ecological models in hydropower dam impact assessment that should be taken into consideration (Table 7.1). Table 7.1 summarizes the variables that were used in the reviewed papers (chapter 2) and key variables that were found in our study (chapter 4, 5, 6). Various models have been developed to get insights in the diverse potential impacts of dams. According to the objectives of the models, model developers can select different types of environmental inputs, ecological assessment levels (species, communities or ecosystems) and scales (temporal and spatial) to develop models according to their specific objectives.

Table 7.1 Checklist for hydropower dams impact assessments

Data collection
Abiotic characteristics
- Elevation
- Rainfall/ precipitation
- Flow
- Velocity
- Depth
- Substrate
- Water temperature
- Turbidity
- Conductivity
- DO
- pH
- Nutrients
- Chlorophyll a
- Toxicants
- BOD
- COD
- Total organic carbon
Biotic components
- Vertebrates
- Invertebrates
- Macrophytes
- Phytoplankton
Assessment levels
- Species
- Communities
- Ecosystems
Sources of data

- Field data
- Lab results
- Expert knowledge

Assessment scales

Time scales

- Before dam construction
- During dam construction
- During dam operation: day/night, week day/weekend, monthly, seasonally
- After dam removal

Spatial scales

- Upstream of the dam
- Reservoir
- Downstream of the dam

Type of models

- Hydrodynamic models
- Water quality models
- Habitat suitability models
- Foodweb models
- Integrated models

7.4 The use of macroinvertebrates to assess the ecological impact of hydropower dams

In this study, macroinvertebrates were used as bioindicators to assess the impact of the hydropower dams on water quality and biodiversity. The understanding of the relationship between environmental factors and aquatic communities is crucial in order to conserve freshwater biodiversity and sustain ecological integrity (Jun et al., 2016). Freshwater macroinvertebrates are widely used as bioindicators in water quality monitoring and assessment, because they have limited mobility, are sensitive to local disturbances, are able to detect structural changes and habitat loss, and different species have different degrees of tolerance towards pollution (Molozzi et al., 2012). Using macroinvertebrates as bioindicators to assess the ecological impacts of hydropower dams has several advantages. Firstly, ecological assessments based on macroinvertebrates are generally simple and cheap to implement (Marchant et al., 2006). The construction of hydropower dams is booming in certain developing countries, however physico–chemical methods require good laboratories with expensive analytical equipment . In addition, hydropower dams are often constructed in remote areas, as such it is not practical to transport water quality samples to the lab for analysis due to the long distances. Moreover, some of these developing countries have limited technical and financial resources for the regular monitoring and regulation of environmental issues. Therefore, cost-effective monitoring programs are needed (Dominguez-Granda et al., 2011b). Secondly, sampling macroinvertebrates and related assessment methods have a long tradition and can count on many standard approaches. Therefore, the results can easily be shared and compared between countries and reported to policy makers. Thirdly, biological indices based on macroinvertebrates have many advantages for testing water quality (Zamora-Muñoz et al., 1995). It has been shown that macroinvertebrate-based indices reflect water quality conditions better compared to physico-chemical indices alone (Sharifinia et al., 2016). A combination of several diversity and biotic indices could take advantages of the strengths of each and develop a more complete understanding of community structure (Hooper et al., 2005). The results from this study also indicated that macroinvertebrate communities accurately reflect factors such as stream velocity and habitat condition (e.g. presence of water hyacinth) and are thus suitable for dam impact assessment.

The results from TITAN showed that there was discrimination between the macroinvertebrate communities in stagnant water (Daule-Peripa hydro reservoir) and those in running waters. Dams shift macroinvertebrates composition from lentic to lotic fauna in the reservoir in relation to the change of stream velocity. For example, the Notonectidae and Dugesiidae exhibited a general preference for lotic habitats, while the Helicopsychidae and Leptophlebiidae were found at sites with lentic habitats. This information is of particular importance for river management practices and biodiversity conservation.

Nevertheless, certain limitations should be taken into account. Firstly, ecological assessment based on macroinvertebrate distribution can be labour intensive and time-consuming at the identification stage (Marchant et al., 2006).

However, it was indicated that only less than six percent of the information was lost by identifying taxa to family (or genus) level, as opposed to species level, and that quantifying the abundance of taxa provided greater resolution for pattern interpretation than simply noting their presence/absence (Marshall et al., 2006). Therefore, identification to family level is recommended. Secondly, many macroinvertebrate taxa exhibit seasonal life cycles to take advantage of optimal environmental conditions, or avoid unfavorable conditions for certain environmental variables such as temperature, water flow status, and food availability (Johnson et al., 2012). The taxon abundance variations of macroinvertebrates related to their life-cycle seasonalities can potentially confound bioassessments (Johnson et al., 2012). However, a previous neotropical study in the Guapimirim River, Brazil, found that the faunal composition was not influenced by seasonal changes, but the densities of macroinvertebrates were negatively influenced by rainfall (Buss et al., 2004). In addition, a case study on stream macroinvertebrates stressed that similar results and biological information in daytime/nighttime data (Guareschi et al., 2016). Therefore, macroinvertebrates are not able to quantify the impact over short time scales (e.g. hydropeaking, day/night pattern); in this way, macroinvertebrates should be used to assess the long-term impact of the dam on ecological water quality and ecosystems.

7.5 Recommendations for river management

The results highlighted the strong association between macroinvertebrate communities and physical-chemical and habitat characteristics. The hydropower dam is a key driver for the change of land use. Agricultural activities, urbanization and untreated wastewater can lead to increased nutrient concentrations, higher conductivity and enhanced chlorophyll concentrations in water. Damming resulted in river fragmentation and altered stream velocity. Therefore, in order to conserve biodiversity and improve water quality in Ecuadorian rivers, some practical aspects should be taken into consideration.

Firstly, sustainable land management is required. It has been shown that land use has a strong impact on chemical quality of water and biological community across the river (Ometo et al., 2000). Land use within the Guayas River basin consists mostly of arable land, plantations (banana, rice, maize, African oil palm

Elaeis guineensis and cacao), urban area and semi-urban areas. The Portoviejo River basin is one of the most productive farming regions in Ecuador, with production of bananas, mangoes and other tropical fruits, tomatoes, onions, peppers, coffee, and especially cattle and fish (http://www.gutenberg.us 2016). Intensive agriculture often involved mono-cropping with high inputs of fertilizers, pesticides and herbicides, which may lead to groundwater contamination and biodiversity degradation (Aude et al., 2003). It has been shown that pesticides used in banana production may enter watercourses and bring ecological risks to aquatic ecosystems (Castillo et al., 2006). Therefore, sustainable land use management can improve river water quality by nutrient limitation (Damanik-Ambarita et al., 2016a) and reduce pesticide run-off into ecosystems. However, further studies regarding the impact of land use on water quality and macroinvertebrate communities distributions are required.

Secondly, damming caused changes in the natural flow regime and could negatively affect stream ecosystems. The first evident change was the flow velocity. Stream velocity has been shown to be an important parameter affecting the distribution of biological communities. One of the key objectives of government policies is meeting human needs for energy and food while minimizing resource consumption (Pfister et al., 2011). Hydropower generation transforms rivers and their ecosystems by fragmenting channels, altering river flows and thus reducing flow velocity (Renofalt et al., 2010). The stream velocity of the Guayas River basin can be regulated and managed by the Daule-Peripa dam. This dam was constructed for hydropower generation, irrigation, flood protection and drinking water (Arriaga, 1989). This study revealed that flow velocity was one of the major variables influencing macroinvertebrate communities. However, the flow of the river was controlled by the dam for various purposes. During dam operation, the water is being released, thus increasing flow and velocity in the downstream river. Understanding the effect of variability in stream velocity on the macroinvertebrate communities can provide basic information to define environmental flows. Dam managers can manipulate dam operating rules to maintain economic benefits while simultaneously conveying adaptive environmental flows for biodiversity (Arthington et al., 2006). Information about community thresholds, such as the one obtained in this study, can be used to adjust stream velocity levels for Ecuadorian rivers. Acreman and Dunbar (2004) suggested that defining the environmental flow was only one step in river protection;

the natural flow of the river and flow needed to maintain local human needs (e.g. navigation, downstream uses) should also be considered (Acreman and Dunbar, 2004). Both the Guayas and the Portoviejo River basins need to be better explored in order to improve knowledge regarding the impact of damming and other human activities on aquatic ecosystems. To our knowledge, no environmental/ecological flow is required to support aquatic life in the Ecuadorian rivers. However, it is necessary to perform a comprehensive cost-benefit analysis of the dam in order to maximize the 'profit'.

Thirdly, our study showed that water hyacinth cover was an important variable affecting the diversity of macroinvertebrates and ecological water quality in the Daule-Peripa reservoir. In the case of the Daule-Peripa reservoir, an intermediate vegetation cover of water hyacinth was positively related with the diversity of macroinvertebrates. Macrophytes can provide excellent microhabitats that promote the establishment and colonization of macroinvertebrates. Although, the spreading of water hyacinth in both Daule-Peripa and Poza Honda reservoirs has been blamed for serious socio-economic consequences such as obstruction of waterways and reducing hydropower production, the benefit of water hyacinth should be weighed and compared to other impacts on ecosystem services before management actions are initiated. It might be a good approach to keep part of water hyacinth cover to support the macroinvertebrate communities and improving ecological water quality. On the other hand, other areas of water hyacinth cover can be harvested for other purposes such as biogas production for local uses. Water hyacinth is considered as a promising material for fuel ethanol production in tropical countries because of its high availability and high biomass yield (Das et al., 2016). Some studies have explored the possibility of (co-) digestion related to (enhanced) biogas/methane production. For example, the mix water hyacinth with sheep waste was tested in different combinations for anaerobic co-digestion, to generate biogas and at the same time digested sludge that can be used as fertilizer for agricultural applications (Patil et al., 2014). Water hyacinth is a very good biogas producer, but needs pre-treatment to enhance the biogas yield (Patil et al., 2014). The use of water hyacinth for biogas production might be promising approach. In some places, water hyacinth can be harvested and used as feedstock in anaerobic digesters to produce methane and generate renewable energy. Biofuels produced from water hyacinth may be an economic management solution in order to control the water hyacinthin the hydropower reservoir. However, as part of the cost-benefit of this methane/biogas generation, a major challenge of water hyacinth is the transport, as it contains floating structures rich in air, and also high quantities of water, in this way, the quotient solids/volume is relatively low.

Conductivity, chlorophyll a and nutrient concentration are considered as major ecological drivers for the protection of the aquatic ecosystem in Ecuadorian rivers. Conductivity integrates several variables like minerals from pollutant degradation (wastewater) and inorganic pollutants (D’heygere et al., 2003). Nutrients (e.g. NO_3-) get into the water as a result of domestic wastewater discharges, agricultural activities (e.g. application of manure and fertilizer containing NO_3-) and as a result of oxidation of nitrogenous waste products in human and animal excreta (Singh and Sharma, 2014). Therefore, in order to improve the ecological water quality, we emphasize the need of integrated management actions to control the diffuse pollution and future investment in wastewater treatment in order to reduce the pollutant load entering into the river. A potential option to improve the water quality is the investment in wastewater treatment. Cuenca is one of the cities in Ecuador which has treatment plants for sewage; even so, it could process only 9% of the sewage (Proano, 2005). Therefore, water quality can be improved by providing sanitation infrastructure (e.g. waste stabilization ponds and constructed wetlands) in order to treat the wastewater before discharge into the river (Troyer et al., 2016). In Ecuador, water quality issues, aquatic ecosystems and ecosystem services have received very limited attention (Nolivos et al., 2015). Although Ecuador has a legal water framework, there are no minimum flow requirements for dam operators, and no standards are available for chlorophyll a concentration and conductivity. In the Guayas River basin, the lack of an appropriate management structure for maintaining the natural conditions caused losses in biodiversity, eutrophication and reduced water availability (Andres, 2009). Nevertheless, integrated management actions should be taken to protect and restore ecosystems and sustainable hydroelectric development in Ecuador as well as the rest of the world.

7.6 Further research

In case of the Daule-Peripa hydropower reservoir, we found that an intermediate vegetation cover of water hyacinth was positively related to the diversity of macroinvertebrates. However, data on the more detailed impact of water hyacinth coverage on electricity generation is not available. The interrelation between different aspects (e.g. impact of water hyacinth on water quality and macroinvertebrates, impact of water hyacinth on electricity production) is necessary. It supports to gain insight in how to steer and control water and ecosystems in a sustainable manner, considering social, economic and political processes (Goethals, 2013). Therefore, further research should quantify these findings and assess the costs and benefits related to the presence of water hyacinth. Integrating the specific ecological knowledge with economic assessments and public perception would deliver valuable information to identify priority habitats to be targeted for the control of water hyacinth and to prioritize conservation actions in an operative way. Understanding of both ecological aspects and socio-economic dimensions can enhance sustainable river management.

Only turbidity was significantly different between sampling sites with and without water hyacinth (c.f. chapter 4). Nevertheless, turbidity should not be considered as a main factor for the presence of water hyacinth in the Daule-Peripa reservoir, but rather should be considered as an effect. Results from chapter 5 showed that flow velocity is the key variable for the establishment of water hyacinth in the Guayas river basin; water hyacinth was only established at sites with a flow velocity lower than 0.4 m/s. However, there is no clear relationship between the presence of water hyacinth and the other physical-chemical variables (e.g. temperature, conductivity, TDS, pH, chlorophyll a, chlorides, DO). The lack of data of water nutrient levels in the Guayas river basin was a major limitation of this study. Therefore, the cause-effect relationships between the presence of water hyacinth and environmental variables remain unclear. Further studies on cause-effect relationships between water hyacinth coverage and nutrient concentrations are needed in order to have better understanding of the habitat suitability of water hyacinth.

In this study, TITAN revealed thresholds for the change of macroinvertebrate communities along a gradient of stream velocity in the Guayas River basin and conductivity and nitrate-nitrogen in the Portoviejo River basin in the dry season. This is the first publication determining the thresholds on macroinvertebrate communities in this region. Further studies in the wet and dry seasons are needed to obtain a full understanding of how the seasonal differences affect the macroinvertebrate communities. The results may serve as a starting point for future studies, which is requiredto check if the thresholds identified from this case study are consistent and applicable at a larger scale and for other species in the ecosystems.

Hydropower generation can cause both direct as well as indirect detrimental impacts on river systems by altering the water flow pattern and reshaping natural habitats (Chen et al., 2015). However, the impacts of hydropower generation on other taxa in the ecosystems such as fishes, algae, vascular plants and birds in Ecuador remain unclear. The change in natural flow patterns can severely disrupt natural riverine production systems. A case study from the Mekong river basin showed that hydropower generation will reduce fish productivity, as well as biodiversity loss (Ziv et al., 2012). Therefore, it is needed to extensively study the impacts of damming on localized fisheries and agriculture production in Ecuadorian rivers. Furthermore, future research should reveal long-term effects of damming on biodiversity and ecological services of rivers in order to have fuller understanding of the ecological impacts.

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Summary

Hydropower dams have gained high attention as ecological friendly energy resources. During recent years, rivers in Ecuador have been heavily impacted by human activities, especially due to the expansion of hydropower dams. There is an urgent need to assess the ecological impacts of hydropower dams and evaluating the water quality of Ecuadorian rivers, as this information is indispensable for future decision-making and management in relation to hydropower dam development, as well as controlling impacts resulting from anthropogenic disturbances. The aim of this thesis is to gain insight the ecological impacts of the hydropower dams on river systems.

Based on the literature search, we found that the models applied to hydropower dam impact assessment have been used to identify an expanding range of impacts at various spatial and temporal scales. Existing ecological models provide a basis to assess the impact of changing hydrological regimes and water quality on the habitat suitability of fish, macroinvertebrates and algae. Field data and expert knowledge have been used as input variables for hydrodynamic models, water quality models, food web models, habitat suitability models and integrated models to explore the change in water quality and habitat suitability at various scales. Given their numerous strengths and opportunities, using models for hydropower dam impact assessment deserves further exploration to improve the understanding of the different processes affected by hydropower dam development and operation. In general, most studies were based on a very limited number of sampling sites. They often only focused on the assessment of an area near to the dam. Consequently, results reported tend to underestimate or overestimate the impacts of dams on upstream and downstream sites. Therefore, an integrated environmental impact assessment of dams at a different level of detail is required.

The field data were collected in the Guayas River basin in 2013 and the Portoviejo River basin in 2015. Different statistical and modeling methods were applied in order to gain insight into the ecological impacts of hydropower dams on Ecuadorian rivers. The macroinvertebrates and physico-chemical variables were sampled, hydro-morphological conditions and habitat characteristics were recorded

at each sampling site. The Biological Monitoring Working Party-Colombia (BMWP-Colombia) values were calculated to assess the ecological water quality.

In the Daule-Peripa reservoir, a generalized linear model shows that water hyacinth is present at sites with a low turbidity. However, there is no clear relationship between the presence of water hyacinth and temperature, conductivity, TDS, pH, chlorophyll, chlorides, DO, oxygen saturation. Therefore, the cause-effect relationships between the presences of water hyacinth and environmental variables remain unclear. The presence of water hyacinth was an important variable positively affecting the diversity of macroinvertebrates in the Daule-Peripa reservoir.

In the Guayas river basin, the BMWP-Colombia scores indicated that water quality of the sampling sites within the Guayas River basin ranged from good to very bad. Canonical Correspondence Analysis revealed the most important environmental factors influencing the distribution of macroinvertebrate communities in the Guayas river basin were stream velocity, chlorophyll a, conductivity, temperature and elevation. Water hyacinth was only present at sites with flow velocities of lower than 0.4 m/s. Threshold Indicator Taxa ANalysis (TITAN) was able to discriminate between the macroinvertebrate community related to stagnant water (Daule-Peripa reservoir) and the macroinvertebrate community related to running waters. TITAN also revealed change points for the thresholds defined by stream velocity at 0.03 m/s and 0.4 m/s. 34 taxa (63%) were considered as the reliable indicators of stream velocity, of which 20 taxa are indicated as indicators of high stream velocity and 14 taxa as indicators of low stream velocity in the Guayas river basin.

In the Portoviejo river basin, the BMWP-Colombia scores indicated that water quality of the sampling sites within the Portoviejo river basin ranged from good to bad. The BMWP-Colombia scores were positively correlated with elevation and stream velocity while negatively associated with temperature, conductivity, chlorophyll, NO_3^-, NO_2^-, TP and TN. However, TITAN could only revealed change points in elevation, conductivity and nitrate-nitrogen concentrations. Change point analysis identified a negative response of macroinvertebrate communities at an elevation of 30 m a.s.l., conductivity values of 1200 µS/cm and nitrate-nitrogen values of 0.5 mg/L. In addition, the change points revealed a positive response of macroinvertebrate taxa at elevations of 62 m a.s.l., conductivity values of 1433

μS/cm and nitrate-nitrogen values lower of 2.3 mg/L. Atyidae, Corbiculidae, Thiaridae, Acari, Baetidae and Leptohyphidae can be considered as indicator taxa to assist in identifying environmental thresholds of the Portoviejo River.

This study investigated some key issues of hydropower generation on Ecuadorian rivers. Dams often severely alter the functioning of river systems, both locally as well as upstream and downstream. This PhD research provides insights into the ecological impacts of hydropower dams, based on research executed in two river basins in Ecuador. The results of this work show the diverse sets of impacts on the physical habitats, chemical water quality as well as plant and invertebrate communities at various spatial scales. The hydropower dams are one of the main drivers causing unfavourable changes in the riverine biota through changes in the flow (translated to stream velocity). Stream velocity is one of the key drivers influencing the distribution of macroinvertebrate communities in the Guayas river basin. It also plays a strong role in the presence of water hyacinth in the reservoir. Water hyacinth provide positive effects for macroinvertebrates. In that way, hydropower dams create both direct and indirect effects on ecosystems. Besides the impacts from damming, Ecuadorian rivers are facing many other pressures such as changes in land use and intensification of agriculture. Our results pointed out the importance of multiple factor assessments at various spatial scales for better understanding impacts of hydropower generation on river water quality and biodiversity. Consequently, this research is highly valuable for environmental impact assessment studies related to the construction and operation of hydropower dams and necessary mitigating management actions.

Samenvatting

Waterkrachtcentrales kregen de voorbij jaren veel belangstelling als ecosysteemvriendelijke manieren om elektriciteit op te wekken. De voorbije jaren werden verschillende waterkrachtcentrales gebouwd in Ecuador, en blijken deze toch een duidelijke ecologische impact teweeg te brengen, zoals ondermeer de sterke ontwikkeling van drijvende waterplanten op de stuwmeren. Vandaar dat er dringende vragen rijzen om deze ecologische impacten nader en ook kwantitief te bepalen, gezien dergelijke informatie onmisbaar is om goede beslissingen te nemen inzake de verdere uitbouw van energie-opwekking via waterkrachtcentrales. Deze thesis heeft als doel om ecologische inzichten te verwerven inzake de ecologische impact van waterkrachtcentrales op riviersystemen.

Op basis van internationale literatuur kon worden vastgesteld dat heel wat modellen reeds in het verleden werden ontwikkeld en gebruikt om de diverse en toenemende set van toestandswijzigingen van rivieren te beschrijven en voorspellen op verschillende ruimtelijke en temporele schalen. Bestaande modellen geven een basis om de impact van wijzigende hydrologische regimes, waterkwaliteit en habitatgeschiktheid voor vissen, invertebraten en algen te beoordelen. Zowel gegevens als expertkennis werden gebruikt als input voor het ontwikkelen van modellen die hydrodynamische processen, waterkwaliteit, voedselwebben en habitatgeschiktheid beschrijven op verschillende ruimtelijke en temporele schalen. Gezien hun talrijke sterkten en mogelijk opportuniteiten voor modellen in de toekomst, verdienen dergelijk modelleerinstrumenten meer aandacht om de effecten van waterkrachtcentrales te beschrijven en analyseren.

In 2013 en 2015 werden de veldgegevens voor deze studies respectievelijk verzameld in de stroombekkens van de Guayas en Portoviejo rivieren. Verschillende statistische gegevensanalysemethoden en modellen werden gebruikt om inzicht te verwerven in de ecologische impacten van waterkrachtcentrales in Ecuadoriaanse rivieren. Daarnaast werd de Biological Monitoring Working Party-Colombia (BMWP-Colombia) gebruikt als een kwaliteitsindex om de waterkwaliteit a.d.h.v. invertebraten te beoordelen.

Via Generalized Linear Models (GLMs) kon de relatie aangetoond worden tussen lage turbiditeitswaarden en de aanwezigheid van waterhyacint stroomopwaarts van de Daule-Periba dam in het Guayas stroombekken. De aanwezigheid van deze waterplanten had bovendien op zijn beurt een grote invloed op de samenstelling van de invertebratengemeenschap. In het algemeen schommelde de BMWP-Colombia van zeer slecht tot goed in dit stroombekken. Via Canonische Correspondentie-analyse kon bovendien aangetoond worden dat stroomsnelheid, chlorofielconcentratie, conductiviteit, temperatuur en hoogteligging van de rivier in sterke mate een verklaring konden geven voor de aangetroffen invertebratenfamilies. Aansluitend kon via Threshold Indicator Taxa ANalysis (TITAN) analyses een onderscheid gemaakt worden tussen invertebratenfamilies die typisch voorkomen bij lage stroomsnelheden (stroomopwaarts van de dam) en hoge stroomsnelheden. In het Portoviejo stroombekken, schommelden de BMWP-Colombia scores van slecht tot goed. In dit bekken kon via de gegevens te analyseren via TITAN aangetoond worden wat omslagpunten waren voor hoogteligging, conductiviteit en nitraatstikstof in relatie tot invertebratenfamilies.

Deze studie analyseerde dus enkele sleutelelementen inzake gekoppelde ecologische wijzigingen van waterkrachtcentrales in rivieren inzake hydromorfologie, waterkwaliteit en biologie. Op die manier is deze studie een belangrijke basis om de ecologische impacten stroomopwaarts- en stroomafwaarts te analyseren, tevens in relatie tot andere impacten, en vormt ze en basis voor verder onderzoek m.b.t. ecologische impacten van waterkrachtcentrales, evenals de vertaling naar de praktijk toe voor milieu-impactstudies en het duurzamer inplanten en beheren van deregelijke systemen.

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Thi Hanh Tien Nguyen, Pieter Boets, Koen Lock, Minar Naomi Damanik Ambarita, Marie Anne Eurie Forio, Peace Liz Sasha Musonge, Natalija Suhareva, Elina Bennetsen, Gert Everaert, Luis Elvin Dominguez Granda, Thu Huong Thi Hoang and Peter Goethals. Ecological water quality and Threshold responses of macroinvertebrate communities to stream velocity in the Guayas River basin (Ecuador). National Symposium for Applied Biological Sciences, February 7th, Leuven, Belgium

Thi Hanh Tien Nguyen, Pieter Boets, Koen Lock, Marie Anne Eurie Forio, Minar Naomi DamanikAmbarita, Gert Everaert, Wout Van Echelpoel, Jana Van Butsel, Juan Antonio Dueñas Utreras, Luis Elvin Dominguez Granda, Thu Huong Thi Hoang, Peter L.M. Goethals. Use of Threshold Indicator Taxa ANalysis method to detect macroinvertebrate community responses to the environmental gradients in Ecuadorian rivers. International Congress on Ecological Modelling and Software, 10^{th}-14^{th} July 2016, Toulouse, France.

Tien Hanh Thi Nguyen, Pieter Boets, Koen Lock, Minar Naomi Damanik Ambarita,Marie Ane Eurie Forio, Peace Sasha, Luis Elvin Dominguez-Granda, Thu Huong Thi Hoang, Gert Everaert, Peter L.M. Goethals. Effect of water hyacinth on water quality and macroinvertebrate diversity in a tropical reservoir. Pawees-INWEPF International Joint Conference 2015 (PIJIC2015): Solutions for sustainable water and environmental management. 19^{th} – 21^{th} August 2015. University Putra Malaysia, Selangor, Malaysia.

Thi Hanh Tien Nguyen, Gert Everaert, Hoang Thi Thu Huong, Peter L.M. Goethals. Overview impact of hydropower development to aquatic resources and environmental aspects in Vietnam.International Fisheries Symposium (IFS2012), 6^{th}-8^{th} December 2012, Can Tho, Vietnam.

Educational activities

Tutor of master thesis

Natalija Semjonova (2013 – 2014). Ecological assessment of the Guayas river basin in Ecuador.Ghent University, Master of Sciences in environmental sanitation. Ghent University. Promotor: Prof. dr. ir. Peter Goethals Tutor: Nguyen Thi Hanh Tien, Gert Everaert and Elina Bennetsen

Promotor of bachelor theses

Nguyen Thi Phuong Tuoi (2010). Monitoring the exploitation of aquatic resources in Son La hydropower reservoir in Phu Yen District- Son La province. Vietnam National University of Agriculture, Bachelor of Aquaculture.

Hoang Kim Dinh (2011). Monitoring the exploitation and using aquatic resources and potential for aquaculture development in Dakrong district, Quang Trị province.Vietnam National University of Agriculture, Bachelor of Aquaculture.

Grants

2011-2015: Scholarship of Belgian government (BTC) for 48 months mixed PhD at Ghent University, Belgium.

2007-2009: Scholarship of Belgian government (VLIR-UOS, Flemish Interuniversity Cooporation, Belgium) for two-year Master of Aquaculture, Ghent University, Belgium.

2001-2004: Scholarship of Asian Institute of Technology for Bachelor Degree for Joint program between Asian Institute of Technology, Vietnam National University of Agriculture and Research Institute for Aquaculture No.1.

Appendices

Appendix 3.1

SAMPLING PROTOCOL: SITE DESCRIPTION

- Site Name:
- Time and date:
- Sample ID:
- Investigator:

Stream name/lake	
Type of watercourse	River Lake
Coordinates	
Altitude of sampling sites [m.a.s.l]	
Photos of the sampling location (numbering the photos) - Downstream - Upstream - Left bank - Right bank - Substrate	

Description of sites (exceptional, weather conditions, main interruption, …)

Land use of the bank top (Estimate at both banks for the stretch of 100m * 10m)

Type of land use	**% on left bank**	**% on right bank**
forests		
arable land		
residential areas		
road, paths		
urban area		
quarrying or mining		
orchard		
other		

Shading

partly shaded, limited stretch <33%	
partly shaded, longer stretch 33-90%	
partly shaded, whole stretch >90%	
completely shaded, limited stretch >33%	
completely shaded, longer stretch 33-90%	
completely shaded, whole stretch >90%	

Presence of macrophytes (% of the bed covered by Macrophytes) (Estimate area cover at the littoral zone of 100m * 10m)

	Submerged aquatic macrophytes	**Emerged aquatic macrophytes**	**Floating aquatic macrophytes**
Contigous/Interrupted			
Abundant = 75-100%			
Common = 50-75%			
Frequent = 25-50%			
Occasional = 5-25%			
Rare = 1-5%			
Invisible			

River morphology

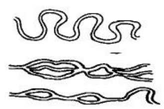
25 valley form

☐ canyon
☐ V-shaped valley
☐ trough

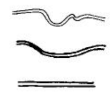
☐ meander valley
☐ U-shaped valley
☐ plain floodplain

26 channel form

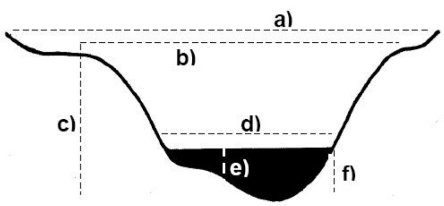
☐ meandering
☐ braided
☐ anabranching



☐ sinuate
☐ constrained (natural)
☐ constrained (artificial)



27 cross section



a) width of floodplain [m] ______
b) flood prone area width [m] ______
c) entrenchment depth [m] ______
d) average stream width [m] ______
e) mean depth water body [m] ______
f) maximum depth water body [m] ______

Free drawing

variation in width

3 cross sections

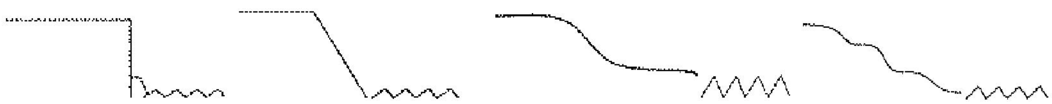
Section 1:	a) [m]____________	b) [m]____________	c) [m]____________
	d) [m]____________	e) [m]____________	f) [m]____________
Section 2:	a) [m]____________	b) [m]____________	c) [m]____________
	d) [m]____________	e) [m]____________	f) [m]____________
Section 3:	a) [m]____________	b) [m]____________	c) [m]____________
	d) [m]____________	e) [m]____________	f) [m]____________

Bank

erosion	Absent/Limited/Abundant
curvature erosion	Absent/Limited/Abundant
width-	
erosion	Absent/Limited/Abundant

Profile of the bank

Vertical steep (>45°) gradually not trampled composite not trampled



Stream Depth

(Measure the depth across the stream, from right bank to left bank, the measure should be done at approximately 1/6, 2/6, 3/6, 4/5 and 5/6 of the way across the stream, 5 measurements)

Section 1

M1	M2	M3	M4	M5

Section 2

M1	M2	M3	M4	M5

Section 3

M1	M2	M3	M4	M5

Variation in flow

absent

at human constructions

low

moderate

high

Sludge layer

invisible	absent	<5cm	5 - 20 cm	> 20 cm

Dead wood

twigs d<3cm	branches 3-30 cm	branch >30 cm
Absent	Absent	Absent
Limited	Limited	Limited
Abundant	Abundant	Abundant

Current Velocity

(Should be measured at the same location where the depth measurements were taken)

Section 1

S1	S2	S3	S4	S5
B1	B2	B3	B4	B5

Section 2

S1	S2	S3	S4	S5
B1	B2	B3	B4	B5

Section 3

S1	S2	S3	S4	S5
B1	B2	B3	B4	B5

Mineral substrates (% of the bed covering)

%	0	0-20	20-40	40-60	60-80	>80
Invisible						
Boulder (D>256mm)						
Cobble (D=64-256mm)						
Gravel (D=2-64mm)						
Sand (D=0.062-2mm)						
Silt (D=4-62 um)						
Clay (D=0.24-4um)						

Pool/Riffle class

Class 1 Pool-riffle pattern is (nearly) pristine: extensive sequences of pools and riffles.	**Class 2** Pool-riffle pattern is well developed: high variety in pools and riffles.
Class 3 Pool-riffle pattern is moderately developed: variety in pools and riffles but locally.	**Class 4** Pool-riffle pattern is poorly developed: low variety in pools and riffles.
Class 5 Pool-riffle pattern is absent: uniform pool-riffle pattern.	**Class 6** Pool-riffle pattern is absent due to structural changes: uniform pool-riffle pattern due to reinforced bank and bed structures.

Appendix 4.1

Call: glm(formula = Water_hyacinth ~ Turbidity, family = binomial, data = Data_dam)

Deviance Residuals:

Min	1Q	Median	3Q	Max
-1.8578	-0.3930	0.3955	0.6790	1.4323

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	7.6551	3.1163	2.457	0.0140 *
Turbidity	-1.5468	0.6856	-2.256	0.0241 *

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

(Dispersion parameter for binomial family taken to be 1)

Null deviance: 35.165 on 27 degrees of freedom

Residual deviance: 24.414 on 26 degrees of freedom

AIC: 28.414

Number of Fisher Scoring iterations: 5

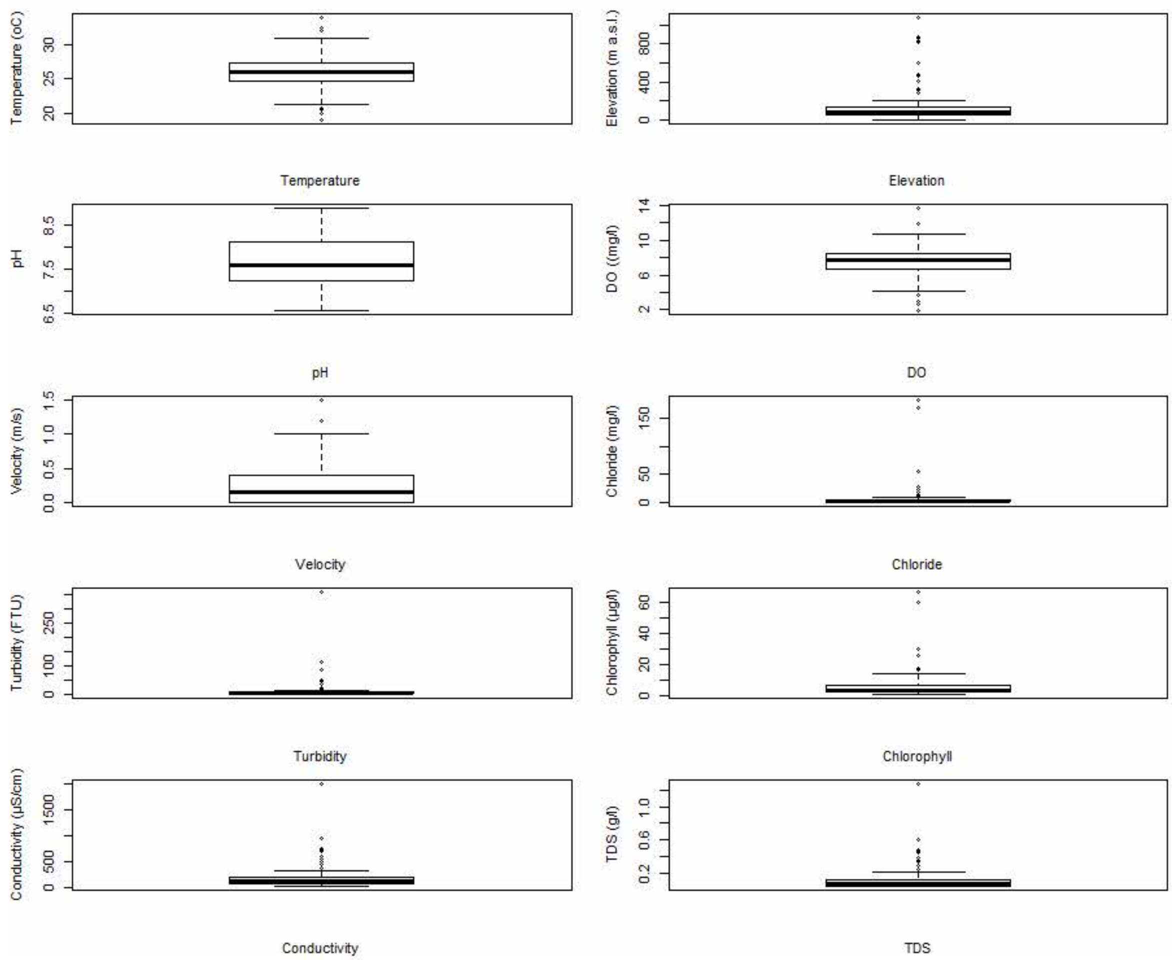
Appendix 4.2

Overview of the different macroinvertebrate taxa found in the samples with indication of the average abundance per vegetation cover class and the percentage of sites where the taxon was found within a certain vegetation cover class (0=absent, 1=1-5%, 2=5-25%, 3= 25-50%, 4 = 50-75%, 5 = 75-100%).

	Average abundance class 0	%sites class 0	Average abundance class 1	%sites class 1	Average abundance class 3	%sites class 3	Average abundance class 4	%sites class 4	Average abundance class 5	%sites class 5
Acari	14.1	81.8	77.8	100.0	12.5	100.0	353.0	100.0	33.5	81.8
Aeshnidae	0.2	18.2	0.0	0.0	1.5	50.0	0.0	0.0	0.5	45.5
Ampullariidae	0.2	9.1	2.0	40.0	9.0	50.0	0.0	0.0	0.0	0.0
Ancylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	9.1
Baetidae	2.3	36.4	3.2	40.0	5.0	100.0	8.7	100.0	5.2	72.7
Caenidae	0.0	0.0	0.0	0.0	0.5	50.0	0.7	66.7	1.2	54.5
Cambaridae	0.3	18.2	0.0	0.0	2.0	50.0	0.0	0.0	0.5	27.3
Ceratopogonidae	0.0	0.0	0.0	0.0	0.5	50.0	0.0	0.0	0.0	0.0
Chaoboridae	0.1	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	122.2	81.8	39.6	80.0	309.5	100.0	110.7	100.0	174.6	90.9
Coenagrionidae	0.1	9.1	0.2	20.0	2.5	100.0	0.7	33.3	6.1	81.8
Corixidae	6.3	27.3	0.0	0.0	0.5	50.0	0.0	0.0	0.0	0.0
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	1.0	33.3	0.5	36.4
Dugesiidae	3.9	27.3	22.0	100.0	0.5	50.0	5.0	100.0	16.0	81.8
Dytiscidae	0.0	0.0	0.2	20.0	1.5	50.0	0.0	0.0	0.4	18.2
Gerridae	1.8	45.5	1.2	40.0	0.0	0.0	1.0	33.3	1.4	27.3
Glossiphoniidae	10.1	72.7	2.8	40.0	20.5	50.0	0.0	0.0	1.6	45.5
Hyallelidae	3.8	36.4	7.2	80.0	102.5	50.0	10.0	100.0	45.1	100.0
Hydrophilidae	0.0	0.0	0.2	20.0	0.0	0.0	0.0	0.0	0.1	9.1
Hydroptilidae	0.0	0.0	0.2	20.0	0.0	0.0	0.0	0.0	0.1	9.1
Libellulidae	0.8	36.4	0.4	40.0	6.0	100.0	5.7	66.7	10.3	100.0
Limoniidae	0.2	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lymnaeidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	27.3

Mesoveliidae	0.1	9.1	2.4	80.0	13.5	50.0	0.0	0.0	0.5	27.3
Naucoridae	0.0	0.0	0.2	20.0	0.0	0.0	0.3	33.3	0.2	18.2
Notonectidae	2.8	27.3	6.6	60.0	36.0	50.0	2.7	33.3	2.3	45.5
Physidae	0.0	0.0	0.0	0.0	0.5	50.0	0.0	0.0	0.0	0.0
Planorbidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	18.2
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	9.1
Thiaridae	125.5	36.4	0.0	0.0	0.5	50.0	0.0	0.0	0.0	0.0
Tubificidae	0.3	18.2	0.8	20.0	0.5	50.0	3.0	66.7	0.9	45.5
Veliidae	0.0	0.0	0.0	0.0	0.5	50.0	0.0	0.0	0.5	9.1

Appendix 5.1: Boxplots reflect sample variabilities of all measured variables in the Guayas river basin. Bold horizontal lines represent median, boxes represent interquartile ranges (25–75% percentiles) and range bars show maximum and minimum values, small black dots show outliers.



Appendix 5.2: Spearman's rank correlation matrix of the 10 environmental variables of the Guayas river basin dataset (n=120). Correlation coefficients with an absolute value of at least 0.70 are marked in bold.

	Temperature	Conductivity	TDS	pH	Chlorophyll	Chloride	DO	Turbidity	Velocity	Elevation
Temperature	1									
Conductivity	-0.11	1								
TDS	-0.18	**0.95**	1							
pH	-0.22	0.43	0.38	1						
Chlorophyll	0.58	-0.24	-0.25	-0.35	1					
Chloride	0.24	0.44	0.47	0.15	0.24	1				
DO	-0.08	-0.05	-0.07	**0.75**	-0.13	-0.01	1			
Turbidity	0.43	0.03	0.03	-0.2	0.59	0.24	-0.22	1		
Velocity	-0.62	0.34	0.35	0.57	-0.69	-0.08	0.26	-0.33	1	
Elevation	-0.43	-0.31	-0.33	0.29	-0.26	-0.39	0.39	-0.42	0.31	1

Appendix 5.3: Threshold Indicator Taxa Analysis of individual taxa in response to stream velocity (m/s) in the Guayas River basin. Taxa are listed in alphabetic order. ienv.cp—environmental change point for each taxon based on IndVal maximum; zenv.cp—environmental change point for each taxon based on z maximum; freq—number of times each taxon occurred in the data set; maxgrp—1 if z- (negative response); 2 if z+ (positive response); IndVal-Dufrene and Legendre 1997 IndVal statistic, scaled 0-100% (with 100 indicating a taxon that occurred in all of the samples above or below a change point value and in none of the samples on the other side of the change point); obsiv.prob—the probability of obtaining an equal or larger IndVal score from random data; (number of random IndVals > = observed IndVal)/ numPerm); zscore—IndVal z score; 5%, 10%, 50%, 90%, 95%—change point quantiles among bootstrap replicates; purity—proportion of replicates matching observed maxgrp assignment; reliability—proportion of replicate obsiv.prob values < = 0.05; z.median—median score magnitude across all bootstrap replicates; filter—logical (if >0) indicating whether each taxa met purity and reliability criteria, value indicates maxgrp assignment. Negative and positive indicators are shown in bold.

Taxa	Shortcode	ienv.cp	zenv.cp	freq	maxgrp	IndVal	obsiv.prob	zscore	5%	10%	50%	90%	95%	purity	reliability	z.median	filter
Acari	**ACARI**	**0.01**	**0.01**	**56**	**1**	**65.04**	**0.001**	**5.45**	**0**	**0**	**0**	**0.03**	**0.08**	**0.99**	**1**	**6.06**	**1**
Aeshnidae	**AESHNIDA**	**0**	**0**	**11**	**1**	**28.95**	**0.001**	**8.76**	**0**	**0**	**0**	**0.02**	**0.03**	**1**	**0.99**	**8.58**	**1**
Ampullariidae	AMPULLAR	0	0.1	6	1	10.91	0.011	3.45	0	0	0	0.11	0.11	0.99	0.90	4.68	0
Ancylidae	**ANCYLIDA**	**0.4**	**0.4**	**13**	**2**	**26.57**	**0.001**	**6.24**	**0.35**	**0.35**	**0.4**	**0.47**	**0.5**	**0.99**	**0.98**	**6.38**	**2**
Baetidae	BAETIDAE	0	0.03	64	1	42.04	0.041	2.02	0	0	0	0.4	0.5	0.86	0.79	2.91	0
Belostomatidae	BELOSTOM	0	0	6	1	12.26	0.016	3.07	0	0	0.15	0.2	0.22	0.96	0.66	2.87	0
Caenidae	**CAENIDAE**	**0**	**0**	**12**	**1**	**30.33**	**0.001**	**8.93**	**0**	**0**	**0**	**0.02**	**0.03**	**1**	**1**	**8.86**	**1**
Calopterygidae	**CALOPTER**	**0.45**	**0.42**	**14**	**2**	**37.8**	**0.001**	**9.22**	**0.28**	**0.3**	**0.42**	**0.45**	**0.47**	**1**	**1**	**9.61**	**2**
Cambaridae	**CAMBARID**	**0.05**	**0.06**	**7**	**1**	**14**	**0.002**	**4.79**	**0**	**0**	**0.01**	**0.06**	**0.07**	**0.99**	**0.96**	**5.70**	**1**
Ceratopogonidae	CERATOPO	0	0	22	2	22.22	0.077	1.57	0	0	0.05	0.47	0.5	0.67	0.64	2.23	0
Chironomidae	**CHIRONOM**	**0.03**	**0.01**	**100**	**1**	**81.75**	**0.001**	**4.01**	**0**	**0**	**0.02**	**0.05**	**0.07**	**0.99**	**0.99**	**4.01**	**1**
Coenagrionidae	COENAGRI	0	0	50	1	47.22	0.014	3.32	0	0	0.1	0.4	0.5	0.79	0.83	3.04	0
Corbiculidae	**CORBICUL**	**0.04**	**0.11**	**22**	**2**	**24.79**	**0.004**	**3.41**	**0.01**	**0.03**	**0.11**	**0.2**	**0.5**	**0.98**	**0.96**	**4.06**	**2**
Corixidae	CORIXIDA	0.72	0.72	28	2	48.62	0.017	3.89	0	0	0.6	0.72	0.72	0.88	0.80	3.83	0
Corydalidae	**CORYDALI**	**0.65**	**0.65**	**11**	**2**	**57.38**	**0.001**	**8.89**	**0.15**	**0.15**	**0.65**	**0.8**	**0.9**	**1**	**1**	**9.17**	**2**
Coryphoridae	**CORYPHOR**	**0.72**	**0.4**	**7**	**2**	**21.88**	**0.001**	**7.55**	**0.35**	**0.37**	**0.42**	**0.75**	**0.77**	**0.99**	**0.99**	**8.42**	**2**
Crambidae	**CRAMBIDA**	**0.6**	**0.6**	**13**	**2**	**60.22**	**0.001**	**11.14**	**0.2**	**0.32**	**0.47**	**0.6**	**0.65**	**1**	**1**	**10.42**	**2**
Culicidae	**CULICIDA**	**0.03**	**0.03**	**14**	**1**	**23.1**	**0.001**	**5.06**	**0**	**0**	**0**	**0.04**	**0.05**	**0.99**	**0.97**	**5.52**	**1**
Dugesiidae	**DUGESIID**	**0**	**0**	**33**	**1**	**56.91**	**0.001**	**7.51**	**0**	**0**	**0**	**0.01**	**0.02**	**0.99**	**0.99**	**6.79**	**1**
Dytiscidae	**DYTISCID**	**0**	**0**	**13**	**1**	**23.32**	**0.001**	**5.57**	**0**	**0**	**0**	**0.05**	**0.06**	**0.96**	**0.99**	**6.24**	**1**
Elmidae	**ELMIDAE**	**0.72**	**0.72**	**20**	**2**	**81.1**	**0.001**	**10.89**	**0.3**	**0.37**	**0.6**	**0.72**	**0.8**	**1**	**1**	**10.76**	**2**
Gerridae	**GERRIDAE**	**0**	**0.03**	**26**	**1**	**26.41**	**0.004**	**3.84**	**0**	**0**	**0**	**0.2**	**0.25**	**0.99**	**0.95**	**4.75**	**1**

Glossiphoniidae	**GLOSSIPH**	**0**	**0**	**28**	**1**	**37.13**	**0.002**	**5.59**	**0**	**0**	**0**	**0.06**	**0.07**	**0.99**	**0.99**	**6.15**	**1**
Gomphidae	**GOMPHIDA**	**0.4**	**0.09**	**17**	**2**	**25**	**0.001**	**5.6**	**0.06**	**0.07**	**0.1**	**0.4**	**0.47**	**1**	**1**	**6.22**	**2**
Helicopsychidae	**HELICOPS**	**0.72**	**0.37**	**14**	**2**	**32.55**	**0.001**	**8.32**	**0.32**	**0.34**	**0.4**	**0.72**	**0.72**	**1**	**1**	**9.40**	**2**
Hyallelidae	**HYALLELI**	**0**	**0.01**	**26**	**1**	**54.39**	**0.001**	**10.78**	**0**	**0**	**0**	**0.03**	**0.03**	**1**	**1**	**11.00**	**1**
Hydrobiidae	**HYDROBII**	**0.45**	**0.2**	**7**	**2**	**13.73**	**0.001**	**4.71**	**0.17**	**0.2**	**0.3**	**0.47**	**0.5**	**1**	**0.96**	**5.30**	**2**
Hydrophilidae	HYDROPHI	0.26	0.26	13	1	15.37	0.023	2.63	0	0	0.2	0.27	0.3	0.97	0.87	3.23	0
Hydropsychidae	**HYDROPSY**	**0.72**	**0.45**	**31**	**2**	**70.88**	**0.001**	**11.15**	**0.32**	**0.37**	**0.45**	**0.6**	**0.6**	**1**	**1**	**12.26**	**2**
Hydroptilidae	HYDROPTI	0.6	0.04	12	2	12	0.086	1.71	0	0.03	0.2	0.65	0.65	0.90	0.71	2.91	0
Leptoceridae	**LEPTOCER**	**0.75**	**0.1**	**27**	**2**	**37.45**	**0.001**	**5.03**	**0.04**	**0.04**	**0.13**	**0.8**	**0.9**	**1**	**1**	**6.20**	**2**
Leptohyphidae	**LEPTOHYP**	**0.72**	**0.4**	**52**	**2**	**84.56**	**0.001**	**8.94**	**0.14**	**0.2**	**0.32**	**0.5**	**0.60**	**1**	**1**	**9.65**	**2**
Leptophlebiidae	**LEPTOPHL**	**0.6**	**0.42**	**30**	**2**	**69.6**	**0.001**	**11.52**	**0.35**	**0.37**	**0.42**	**0.5**	**0.5**	**1**	**1**	**12.48**	**2**
Libellulidae	**LIBELLUL**	**0**	**0**	**55**	**1**	**58.54**	**0.001**	**5.94**	**0**	**0**	**0**	**0.04**	**0.05**	**0.99**	**0.99**	**5.88**	**1**
Limoniidae	LIMONIID	0.6	0.6	14	2	28.46	0.006	4.39	0.11	0.12	0.42	0.6	0.72	0.97	0.94	5.01	0
Lymnaeidae	LYMNAEID	0.5	0.5	10	1	10.42	0.179	1.06	0	0	0.17	0.5	0.5	0.61	0.39	1.71	0
Megapodagrionidae	MEGAPODA	0.6	0.2	6	2	12.24	0.001	4.38	0.15	0.17	0.2	0.6	0.65	0.99	0.89	4.32	0
Mesoveliidae	**MESOVELI**	**0**	**0.03**	**15**	**1**	**22.18**	**0.002**	**4.13**	**0**	**0**	**0.01**	**0.26**	**0.27**	**0.99**	**0.97**	**4.85**	**1**
Naucoridae	**NAUCORID**	**0.72**	**0.45**	**33**	**2**	**44.95**	**0.001**	**5.6**	**0.05**	**0.08**	**0.46**	**0.72**	**0.72**	**1**	**1**	**6.94**	**2**
Notonectidae	**NOTONECT**	**0**	**0**	**17**	**1**	**38.91**	**0.001**	**8.74**	**0**	**0**	**0**	**0.03**	**0.07**	**1**	**1**	**9.09**	**1**
Perlidae	**PERLIDAE**	**0.65**	**0.5**	**12**	**2**	**42.53**	**0.001**	**10.29**	**0.45**	**0.47**	**0.5**	**0.65**	**0.65**	**1**	**0.99**	**11.33**	**2**
Philopotamidae	**PHILOPOT**	**0.65**	**0.07**	**17**	**2**	**22.76**	**0.006**	**3.28**	**0.01**	**0.03**	**0.41**	**0.77**	**0.9**	**0.99**	**0.99**	**4.54**	**2**
Physidae	PHYSIDAE	0.77	0.4	9	2	13.44	0.026	2.56	0	0	0.4	0.8	0.9	0.79	0.80	3.73	0
Planorbidae	PLANORBI	0	0.5	8	1	8.33	0.248	0.73	0	0	0.35	0.45	0.5	0.40	0.39	1.76	0
Platystictidae	**PLATYSTI**	**0.75**	**0.42**	**11**	**2**	**21.89**	**0.001**	**5.21**	**0.01**	**0.03**	**0.42**	**0.6**	**0.72**	**1**	**0.97**	**5.85**	**2**
Pleidae	PLEIDAE	0.15	0.15	7	1	11.29	0.014	3	0	0	0.1	0.17	0.2	0.99	0.87	3.58	0
Polycentropodidae	POLYCENT	0.11	0.11	6	2	9.38	0.028	2.82	0.1	0.1	0.11	0.42	0.55	0.98	0.69	2.99	0
Psephenidae	**PSEPHENI**	**0.72**	**0.45**	**17**	**2**	**53.03**	**0.001**	**11.84**	**0.37**	**0.4**	**0.47**	**0.72**	**0.72**	**1**	**1**	**12.022**	**2**
Simuliidae	SIMULIID	0.45	0.42	8	2	16.47	0.011	3.94	0.07	0.11	0.45	0.65	0.77	1	0.93	5.02	0
Stratiomyidae	STRATIOM	0.5	0.5	7	1	7.29	0.152	0.58	0	0	0.1	0.42	0.45	0.65	0.34	1.75	0
Tabanidae	TABANIDA	0.72	0.72	6	2	42.42	0.002	8.75	0	0.5	0.67	0.72	0.72	0.93	0.91	7.94	0
Thiaridae	THIARIDA	0.77	0.12	36	1	28.26	0.12	1.33	0	0	0	0.2	0.47	0.84	0.54	1.99	0
Tubificidae	TUBIFICI	0.31	0.31	29	1	25.79	0.054	1.69	0	0	0.2	0.32	0.32	0.87	0.73	2.46	0
Veliidae	**VELIIDAE**	**0.45**	**0.37**	**40**	**2**	**61.38**	**0.001**	**7.44**	**0.3**	**0.31**	**0.4**	**0.5**	**0.6**	**1**	**0.99**	**8.06**	**2**

Appendix 5.4 Summary of the canonical correspondence analysis

Partitioning of mean squared contingency coefficient:

	Inertia	Proportion
Total	4.38	1.00
Constrained	1.01	0.23
Unconstrained	3.37	0.77

Accumulated constrained eigenvalues
Importance of components:

	CCA1	CCA2	CCA3	CCA4	CCA5	CCA6	CCA7	CCA8
Eigenvalue	0.45	0.22	0.11	0.08	0.05	0.04	0.03	0.02
Proportion Explained	0.45	0.22	0.11	0.08	0.05	0.04	0.03	0.02
Cumulative Proportion	0.45	0.66	0.77	0.85	0.91	0.95	0.98	1.00

Biplot scores for constraining variables

	CCA1	CCA2	CCA3	CCA4	CCA5	CCA6
Temperature	**-0.80**	-0.40	0.05	-0.42	0.04	-0.04
Conductivity	0.19	**-0.92**	-0.14	0.11	-0.08	0.13
Chlorophyll	**-0.67**	-0.04	0.27	0.30	-0.40	0.12
Cloride	-0.17	-0.42	-0.32	-0.03	-0.17	0.09
DO	0.30	0.09	0.13	-0.48	-0.73	-0.29
Turbidity	-0.37	-0.15	-0.09	-0.04	-0.35	0.76
Velocity	**0.89**	0.11	-0.06	-0.15	0.07	-0.05
Elevation	**0.65**	0.42	0.59	0.00	-0.02	0.00

Appendix 6.1

Spearman's rank correlation matrix of the 16 environmental variables of the Portoviejo River dataset (n=30). Correlation coefficients with an absolute value of at least 0.70 are marked in bold.

	Elevation	Velocity	Temperature	Conductivity	pH	DO	Chlorophyll	Turbidity	BOD_5	NO_3^-	NO_2^-	NH_4^+	TN	oPO_4^{3-}	TP	TOC
Elevation	1															
Velocity	-0.22	1														
Temperature	-0.33	-0.39	1													
Conductivity	**-0.84**	0.05	0.46	1												
pH	0.03	-0.21	0.36	0.1	1											
DO	-0.07	-0.31	0.38	0.11	0.69	1										
Chlorophyll	-0.57	-0.33	0.53	0.58	0.23	0.49	1									
Turbidity	-0.28	0.42	-0.25	-0.23	-0.28	-0.13	-0.15	1								
BOD_5	0.51	-0.51	0.05	-0.3	-0.12	-0.05	0.08	-0.43	1							
NO_3^-	**-0.81**	0.05	0.43	**0.82**	0	-0.08	0.52	-0.06	-0.31	1						
NO_2^-	**-0.78**	0.01	0.41	**0.75**	0.03	0.34	**0.79**	-0.05	-0.18	**0.7**	1					
NH_4^+	-0.08	0.23	-0.16	-0.08	-0.16	-0.22	-0.32	0.37	-0.37	0.08	-0.1	1				
TN	-0.61	-0.06	0.47	0.63	0.1	0.07	0.36	-0.07	-0.21	**0.79**	0.48	0.03	1			
oPO_4^{3-}	**-0.76**	0.34	-0.06	0.54	-0.08	-0.07	0.33	0.38	-0.39	0.52	0.58	0.08	0.22	1		
TP	**-0.73**	0.09	0.14	0.55	-0.04	-0.02	0.58	0.21	-0.13	0.66	**0.7**	-0.07	0.44	0.83	1	
TOC	**-0.86**	0.15	0.48	**0.73**	0.11	0.23	0.65	0.21	-0.41	**0.74**	**0.76**	0	0.68	0.58	0.68	1

Appendix 6.2

List of all families and their tolerance score of macroinvertebrate taxa collected in the Portoviejo River

Taxa	Shortcode	BMWP-Colombia score	No of present	Frequency
Acari	ACARI	-	184	6
Atyidae	ATYIDAE	8	199	6
Baetidae	BAETIDAE	7	181	19
Belostomatidae	BELOSTOM	4	12	9
Calopterygidae	CALOPTER	7	93	11
Cambaridae	CAMBARID	-	16	6
Ceratopogonidae	CERATOPO	5	19	8
Chironomidae	CHIRONOM	2	805	29
Coenagrionidae	COENAGRI	7	124	21
Corbiculidae	CORBICUL	-	247	9
Corydalidae	CORYDALI	6	19	3
Culicidae	CULICIDA	2	1	1
Dryopidae	DRYOPIDA	6	28	3
Elmidae	ELMIDAE	6	8	6
Ephydridae	EPHYDRID	4	1	1
Gelastocoridae	GELASTOC	5	1	1
Gerridae	GERRIDA	-	8	5
Glossiphoniidae	GLOSSIPH	5	8	1
Gomphidae	GOMPHIDA	9	71	17
Haliplidae	HALIPLID	4	13	9
Hydrobiidae	HYDROBII	7	59	1
Hydrophilidae	HYDROPHI	3	29	5
Hydropsychidae	HYDROPSY	7	74	9
Hydroptilidae	HYDROPTI	8	7	4
Lampyridae	LAMPYRID	10	2	1
Leptoceridae	LEPTOCER	8	22	9
Leptohyphidae	LEPTOHYP	7	175	11
Leptophlebiidae	LEPTOPHL	9	61	8
Libellulidae	LIBELLUL	5	231	20
Limoniidae	LIMONIID	3	15	7
Littorinidae	LITTORIN	-	20	1
Lymnaeidae	LYMNAEID	8	1	1
Mysidae	MYSIDAE	-	1	1
Naucoridae	NAUCORID	8	42	12
Nepidae	NEPIDAE	5	8	3
Notonectidae	NOTONECT	5	135	2
Ochteridae	OCHTERID	-	1	1
Palaemonidae	PALAEMON	8	30	4
Perlidae	PERLIDAE	10	3	1
Philopotamidae	PHILOPOT	9	7	1
Physidae	PHYSIDAE	3	2	2
Pleidae	PLEIDAE	6	12	6
Polycentropodidae	POLYCENT	9	5	1
Ptilodactylidae	PTILODAC	10	1	1
Pyralidae	PYRALIDA	7	11	4

Scirtidae	SCIRTIDA	4	4	3
Simuliidae	SIMILIID	7	2	2
Spionidae	SPIONIDA	-	14	2
Stratiomyidae	STRATIOM	3	12	5
Tabanidae	TABANIDA	5	6	3
Thiaridae	THIARIDA	5	5231	17
Tubificidae	TUBIFICI	1	18	6
Veliidae	VELIIDAE	7	56	11