2020

Developing an Odonate-Based Index for Monitoring Freshwater Ecosystems in Rwanda: Towards Linking Policy to Practice through Integrated and Adaptive Management

Erasme Uyizeye

Department of Environmental Studies

DISSERTATION COMMITTEE PAGE

The undersigned have examined the dissertation entitled: Developing an Odonate-Based Index for Monitoring Freshwater Ecosystems in Rwanda: Towards Linking Policy to Practice through Integrated and Adaptive Management, presented by Erasme Uyizeye, candidate for the degree of Doctor of Philosophy, and hereby certify that it is

accepted*.

Committee Chair: Beth A. Kaplin, Ph.D.
Antioch University New England, USA

Committee member: Lisabeth Willey, Ph.D.
Antioch University New England, USA

Committee member: Viola Clausnitzer, Ph.D.
Senckenberg Museum of Natural History Görlitz, Germany.

Defense Date: April 17th, 2020.

Date Approved by all Committee Members: April 30th, 2020.

Date Deposited: April 30th, 2020.

*Signatures are on file with the Registrar's Office at Antioch University New England.

Developing an Odonate-Based Index for Monitoring Freshwater Ecosystems in Rwanda: Towards Linking Policy to Practice through Integrated and Adaptive Management

By

Erasme Uyizeye

A dissertation submitted in partial fulfilment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Environmental Studies

at

Antioch University New England

Keene, New Hampshire

2020

Dedication

I dedicate this dissertation to

my daughter who was born in the midst of this doctoral journey,

my wife who has stayed by my side,

my father for his words of encouragement (1956-1993)

& my mother for her unwavering support and love.

Acknowledgments

First and foremost, I would like to express my sincere gratitude to my advisor, Beth A. Kaplin for her invaluable support and patience all along my doctoral studies and research. I am also very thankful to my committee members, Lisabeth Willey and Viola Clausnitzer for their advice and guidance as I grow as a scholar.

I am very thankful for different sources of funding, the fellowship from the Center for Tropical Ecology and Conservation, research grants from National Geographic Society and the Rufford Foundation. I cannot possibly list and thank everyone enough who supported me throughout this doctoral journey within the confines of this document. Here, I mention just a few, as a sincere token of my appreciation for their support and love. I am very grateful to the entire faculty for contributing to my learning process at Antioch University New England including but not limited to Alesia Maltz, James Gruber, James Jordan, Jason Rhoades, Jean Kayira, Jimmy Karlan and Joy Ackerman, Peter Palmiotto and Rachel Thiet. Also, many thanks to Sarah Bockus, Amy Roy and Susan Weller for their support. I also feel special gratitude to my fellow graduate students, especially Eli, Francis and Karen. I do thank other people who kept me company such as Amanda, Makani, Yves, Jude, Marshall, Enathe and Apollinaire. I extend my thanks to my collaborators, Jens and KD for joining the field work and assisting in specimen identification. I am also thankful to my field assistants, Alphonse, Diane and Thomas.

I am grateful to my immediate and extended family members for all of their support. These include my mother, sisters, parents-in-law and sister in law, Epiphanie, Dinah, Sonia, Anwar, Connie and Adrienne. I extend my deep gratitude to my loving wife Olivia who blessed me with a life of joy in moments when lights of research and writing were off. Most importantly,

I sincerely thank her for the greatest gift of our daughter, Assana, who is just turning one year old at the time I defend this dissertation. I love you all dearly!

Abstract

Worldwide, the decline of biodiversity in freshwater ecosystems is occurring at an alarming rate, due to anthropogenic threats, which directly impact humans in a variety of ways. Freshwater ecosystems occupy an integral part of political, socio-economic and ecological spheres. Integrated Watershed Management (IWM) and Adaptive Management (AM) conceptual frameworks provide an underpinning holistic platform from which to evaluate the performance of policies and actions on the ground in relation to freshwater ecosystem management. I investigate the extent to which environmental policies and practices embrace IWM and AM frameworks in Rwanda. Furthermore, this dissertation develops an odonate-based ecological monitoring tool, referred to as Dragonfly Biotic Index (DBI). The development of this tool involved surveying adult odonates, water physical-chemical variables, habitat characteristics and weather conditions across the six ecological zones of Rwanda. An average of 16 sites per each ecological zone were surveyed in a short rainy season and revisited in a short dry season. This countrywide survey added 25 new odonate species to the national check list, which increased it to 114 species. The abundance of odonates was significantly different between ecological zones and between seasons. The DBI developed here consists of three sub-indices: distribution-based score, sensitivity-based score and threat-based score as per IUCN Red List categories. To validate DBI, I examined its effectiveness in reflecting habitat integrity. This included using DBI to assess the relationship of land uses (agriculture and mining) and environmental, and physical-chemical variables of freshwater ecosystems. DBI values were significantly lower in agricultural and mining sites than their control sites. Also, significant changes in some environmental variables were associated with the two land uses. These included the degradation of riparian vegetation as associated with both agriculture and mining. While agriculture was significantly

associated with higher conductivity, mining exhibited a significant relationship with higher water turbidity and higher sandy substrates than their control sites. In conclusion, not only will DBI enable deeper investigation of the extent to which land uses affect freshwater ecosystems, but also will be instrumental in prioritization for habitats that need crucial conservation. Additionally, this monitoring tool is meant to make data on ecosystem status readily available to facilitate analysis of ecological responses to socio-economic, political and pragmatic interventions. Thus, these data can be used to inform all spheres involved: ecological, political and socio-economic. The use of odonates, which are charismatic insects, will potentially engage and promote citizen-based monitoring. This will ultimately instill pro-environmental attitudes within local communities and set the stage for collaboration between stakeholders.

Keywords: Odonates, dragonflies, biological indicator, biotic index, freshwater ecosystems, monitoring, integrated management, adaptive management, agriculture, mining, Africa, Rwanda.

Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

Freshwater ecosystems are the richest in biodiversity among aquatic ecosystems. They constitute less than 1% of the world's surface, and harbor about 6% of all the world's known species (Dudgeon et al., 2006; Martens, 2010). These species are declining at an alarming rate due to anthropogenic threats (Dudgeon et al., 2018; Turak et al., 2017). Given the underpinning role of biodiversity in ecosystem functions and services, the loss of species in freshwater ecosystems directly affects humans by impairing these essential services including potable water, food, water for industry, water for agriculture, recreation and navigation (Cunha et al., 2019; Khan et al., 2019; Monteiro-Júnior et al., 2014).

To address the decline in freshwater biodiversity and ecosystem services, effective interventions should be undertaken, especially in areas of high vulnerability such as African developing countries (Holland et al., 2012). Effective interventions require deep and broad understandings of threats and their effects in order to establish efficient ecosystem management. Maintaining timely and well informed decision making and management can be nurtured through regular assessments of the state of ecosystems (Foley et al., 2015; Teder et al., 2007).

This dissertation documents the development and testing of a bioindication based-tool to improve freshwater ecosystem management and planning in Rwanda. Why did I choose to do this work in Rwanda? Freshwater ecosystems in Rwanda exemplify highly vulnerable ecosystems. This is in part due the high human population that is very dependent on Rwanda's ecosystem services (Danielsen et al., 2005; Dawson et al., 2019). The ecological and socio-economic condition of Rwanda, amplified by the current and predicted effects of climate change, raise the vulnerability even higher (Egoh et al., 2020; Marques et al., 2018; Markovic et al., 2014; Taniwaki et al., 2017). From the ecological perspective, the following two narrative sayings are commonly known to describe Rwanda: *Rwanda is the heart of Africa*, and *the country of a thousand hills* (Campioni et al., 2012; Wyss, 2006). Describing Rwanda as the heart of Africa refers to its location and hydro-ecological function for the continent. The country is located in a biodiversity hotspot of East Africa (the Albertine Rift region), seated within the great lakes region, and constitutes part of the upstream catchments for two of the biggest rivers on the African continent, Nile and Congo Rivers (Abtew et al., 2019).

The country of a thousand hills reflects the diversity in ecosystems caused by broad elevation changes. The highest elevation is located in the northwestern part of the country where afroalpine and afromontane forests thrive (4,507 meters). Towards the furthest south corner of the country, the lower elevation provides for gallery forest at points near Lake Kivu. Rwanda's ecosystems vary also from west to east, where the high elevations of the mountains subside into rolling hills and marshy grassland valleys of the central plateau region. The gradual reduction in slope gradient extends toward the east with an area characterized by warmer savanna bushland. These blend into a landscape with broad river valleys, lakes, and papyrus swamp (Kindt et al., 2011).

Furthermore, not only are Rwanda's diverse ecosystems representative of most of the ecosystems of East Africa (Lowe-McConnell, 2010), Rwanda's size makes such a countrywide study more logistically manageable. This means Rwanda is a good natural laboratory for the application of bioindication in monitoring ecosystems in the region and beyond. The study of bioindication-based monitoring in Rwanda could potentially be applied and subsequently advanced for a standardized bioindication method for the region.

In addition to its natural settings, the socio-economic aspects of Rwanda make it an interesting study case for freshwater ecosystems. Rwanda is unique in Africa since it is the most densely populated on the continent and is experiencing the fastest economic growth (Diao et al., 2014). The rapid economic development and food demands are accompanied by compromises to ecosystems, freshwaters in particular. The building of infrastructure goes side by side with the over-exploitation of natural resources, most of which alter freshwater ecosystems (Dusková & Machácek, 2013). Agricultural intensification has increasingly put pressure on wetlands, as they are the only remaining undeveloped, yet unprotected arable areas in the country (Salmah et al., 2006; Uwimana et al., 2018). These put the country on the high end of the spectrum that measures environmental challenges on the continent, stressing the critical need to monitor these ecosystems in order to be able to tackle the changes and competing interests. These ecological, social and economic perspectives raise the need for a reliable, accurate and precise biological indicator for ecosystem monitoring, with a foundation in the bioindication concept.

Bioindication concept

The definition and use of bioindication has been an evolving concept. According to Asif et al. (2018), the first attempt to define bioindication was in 1980. It was defined as simplification of information from an ecosystem to understand the state of the system as whole. Bioindication was redefined two years later by Steubing (1982) and Zonneveld (1983) who illustrated different scales by which bioindication can be applied, mostly at generic levels.

In its most advanced sense, bioindication is an integrated investigation of various biological responses to varied external factors (Parmar et al., 2016). The biological responses tend to reflect the state of environmental pollution, disturbance or degradation. Also, bioindication can support efforts for foresight of development or change in both the absence and presence of intervention (Markert, 2007; McGeoch et al., 2011; Parmar et al., 2016).

While recent studies on bioindication have provided convincing arguments for the use of bioindication, in its early stage, this concept evoked skepticism among scholars such as Roback and Richardson (1969) who pointed out the ambiguity residing in bioindication. They argued that the presence or absence of any species in a stream does not always indicate the state of water quality. Their explanation was that species occurrence may be due to the random colonization of the species pool in the area studied or a response to the season in which the collection is made. Roback and Richardson's (1969) point was dismissed by Steubing (1982), Toft and Schoener (1983) and many other studies that came after, including the most recent ones, such as Mu et al. (2000) and Perry et al. (2010) who argued that the presence or absence of individual species can be used as a sign of habitat status.

This dissertation harmonizes with post Roback and Richardson's (1969) arguments, given it is based on optimization of precision and accuracy of bioindication (Brito et al., 2018; Salmah et al., 2006; Mangadze et al., 2019). The biological indicator developed in this dissertation considers a wider array of levels, ranging from species and their populations, to their communities. Not only could this be anticipated to elevate the certainty of bioindication of various stressors to specific species, but it could also allow an adequate analysis of the integrated responses of populations and communities as a whole (Brito et al., 2018; Salmah et al., 2006; Mangadze et al., 2019).

The use of macroinvertebrates as indicators

Invertebrates, and macroinvertebrates in particular, have commonly been used as bioindicators of aquatic habitats (Brito et al., 2018; Salmah et al., 2006; Mangadze et al., 2019; Theodoropoulos et al., 2020). They can reflect changes in the environment through their responses to stressors, which impact their community structures These can be observed through a range of reactions from species presence/absence ratios to changes in the whole invertebrate community (Kiffer & Marcelo, 2017; Simaika & Samways, 2011). Assessing changes at the community level have been found to be the most appropriate approach to bioindication in the long-term and over a wide space (Mendes et al., 2017; Siddig et al., 2016).

Although the use of macroinvertebrates in habitat assessment is often seen as time intensive, it signals ecological conditions in a cost-effective way. On this basis, the information provided to decision-makers for environmental conservation and remediation is accurate (Mendes et al., 2017). On top of this, bioindication has an added advantage as it may transcend informing policies and reach local communities. This is evident particularly when the indicator organisms are charismatic as they are more likely to be embraced in citizen science. This increases public participation in long-term monitoring. This is particularly true for odonates, biological indicators for freshwater monitoring (Conrad & Hilchey, 2011; Overdevest et al., 2004).

Using adult odonates over benthic macroinvertebrate larvae

Odonates are increasingly being demonstrated as efficient biological indicators. They show advantages over using many other taxa such as fishes, benthic macroinvertebrates and diatoms and aquatic macrophytes (Dalu & Froneman, 2016; Gerson Araujo et al., 2003; Mangadze et al., 2019; Suganuma & Durigan, 2015; Taniwaki et al., 2017). While these approaches have a series of limitations attached to their use, it would be unrealistic to claim a complete substitute that uses odonates over other taxa. Each of the taxa has its own pros and cons depending on the types of ecosystems or objectives of assessment.

Regarding advantages of using macroinvertebrates, there are multiple shortcomings that the use of benthic macroinvertebrates bear. Benthic macroinvertebrates are a broad group of organisms with a huge taxonomic diversity. These include, for example, larvae of caddisflies, dragonflies, mayflies, stoneflies, snails and beetles, each which has numerous species. It is often difficult to reach the lowest taxonomic resolution, the species level, since the larval stage are not morphologically distinctive enough (Brito et al., 2018). This makes the establishment of precise and accurate causal relationships between external factors and the composition of entire invertebrate communities difficult (Simaika & Samways, 2009a; Turak et al., 2017). Additionally, sorting and identification of benthic macroinvertebrates can be time consuming and expensive (Jeanmougin, 2014; Siddig et al., 2016; Dufrene & Legendre, 1997). The use of benthic macroinvertebrates does not represent other habitats outside their specific bodies of water. Also, they are not sensitive to changes in hydro-morphology of bodies of water (Garcia et al., 2012; Golfieri et al, 2016).

The use of odonates fills in these limitations and offers several other advantages. Odonates reflect the impact of environmental change and act as proxies for both aquatic and terrestrial habitats (Remsburg & Turner, 2009). This is due to their amphibiotic life cycle, which means they live in aquatic habitats, for part of their life development, and terrestrial habitats when they become adults (Dutra & Marco, 2015). Their prolonged nymphal phase in aquatic habitats allows odonates to reflect the ecological integrity and habitat heterogeneity of bodies of water (McPeek, 2008).

The use of adults of odonates in habitat assessment and monitoring has been shown to be practical (Mendes et al., 2017). They are relatively easy to identify to species level due to their morphologically distinctive traits between species. This maintains the accuracy in assessment results (Valente-Neto et al., 2016; Vorster et al., 2020; Le Gall et al., 2018). Many other empirical and analytical studies suggest that a positive relationship between invertebrate assemblage and habitat characteristics becomes much clearer when invertebrates are analyzed to the species level. It has been suggested, therefore, that assessment metrics with a finer resolution, such as odonate species are the best approach (Jeanmougin, 2014; Siddig et al., 2016; Dufrene & Legendre, 1997).

Due to the fact that adult odonates are easily observable and conspicuous, odonates can serve as instrumental candidates for rapid habitat assessment of ecological integrity and are particularly valuable for medium to long-term monitoring programs (Siddig et al., 2016). These practices could focus on monitoring just rare species. Assessment could also look at the entire odonate communities. Either way, monitoring odonates Can provide accurate indication of the condition of freshwater habitats. It can also be a pathway to rating restoration and conservation priorities (Mendes et al., 2017; Rodrigues et al., 2016). Furthermore, odonates can offer specific insights about the condition and structure of the aquatic and riparian vegetation types, such as short grasses, tall wetland grasses and shrub (Remsburg & Turner, 2009). Occurrence patterns in odonate assemblages should be, therefore, useful for freshwater habitat assessment (Golfieri et al., 2016).

Odonates as flagships for freshwater habitat preservation

Odonates are the only freshwater insect group that has been systematically assessed on a global scale (Clausnitzer et al., 2009; Clausnitzer, V. & Jödicke, 2004). This global assessment was officially initiated in 2005, when International Union for Conservation of Nature (IUCN) took on its initiative to update the odonates' red-list. This includes assessing odonate distribution and extinction risk following the IUCN guidelines (Clausnitzer et al., 2012). This global assessment and other studies that came later have suggested tremendous declines in odonate populations and species extinction (Butchart et al., 2018; Clausnitzer et al., 2009; Kalkman et al., 2018). The primary causes of these losses include over exploitation of ecosystems, invasive species and impacts of climate change (Taniwaki et al., 2017). It is thought that the decline of odonates correlates with the declining trend of other freshwater biodiversity. The decline of biodiversity in wetlands is up to five times greater than the biodiversity loss in terrestrial ecosystems (Dudgeon et al., 2006).

Ensuring the protection of odonates implies preserving both aquatic and terrestrial habitats, given their amphibiotic dependence to these habitats (Miguel et al., 2017). They can play a flagship role for conservation of other overlooked species in both terrestrial and aquatic habitats(Clausnitzer et al., 2012). Odonates are associated with habitat characteristics of their ecosystems. Some species are dependent on, for example, the presence and stability of emergent seepage for successful reproduction. Odonate presence is linked to the steady provision of groundwater, and they can thus be used to assess the impact of water loss and other activities that reduce the water table. This is also important in selecting habitats that need preservation (Baird & Burgin, 2016; Garcia et al., 2012).

Odonates can be used in selecting habitats that need preservation. This process is usually done through "the complementarity approach". To select a reserve, the complementarity approach strives to take into account as many ecological attributes as possible at a minimum area (Kati et al., 2004). Selecting a reserve on the basis of areas of high richness for just one taxon is rarely representative of other taxa and does not include other important attributes. However, odonates have been suggested as appropriate candidates to address this limitation, given some monitoring indices, such as the Dragonfly Biotic Index (DBI), account for global status as per the IUCN Red List, and endemism, among others. Additionally, the DBI represents the global Red Listed species within a site (Simaika & Samways, 2009b).

Dissertation Outline

This dissertation includes three interconnected empirical chapters, each of which is written in manuscript format. Each chapter includes abstract, introduction, methods, results, discussion, conclusion, references and appendices.

After the general introduction comes chapter 2, which presents a policy and law analysis. Here, the focus is put on exploring how laws and policies in Rwanda are aligned with principles from a hybrid of two frameworks, Integrated Watershed Management (IWM) and Adaptive Management (AM). This chapter is grounded in the notion that wetlands are integral to our watersheds and an important landscape component that plays an instrumental role at the political, socioeconomic and ecological interface. As a product of IWM and AM frameworks, chapter 2 recommends an ecological monitoring approach be used for freshwater habitats.

Chapter 3 develops a Dragonfly Biotic Index (DBI), an ecological monitoring tool for Rwanda (Samways & Simaika, 2014; Simaika & Samways, 2009a; Vorster et al., 2020). This chapter also analyzes differences in odonate assemblages between seasons and across ecological zones at several sites in Rwanda and outlines benchmark sites that can play a seminal role in restoration. Benchmarks are defined from DBI site values, species richness and presence of unique or endemic species.

Chapter 4 applies the DBI developed in chapter 3 and explores its effectiveness in indicating the analogy of agriculture and mining in relation to their effects on freshwater habitat integrity. This chapter also evaluates how changes in environmental and physical-chemical variables are indicated by odonates.

References

Abtew, W., & Dessu, S. B. (2019). *The Nile River and Transboundary Water Rights* (The Grand). Springer, Cham.

Asif, N., Malik, M., & Chaudhry, F. N. (2018). Biogeographie. *Pollution*, *4*(1), 111–118. https://doi.org/10.22059/POLL.2017.237440.296

Baird, I. R. C., & Burgin, S. (2016). Conservation of a groundwater-dependent mire-dwelling dragonfly: implications of multiple threatening processes. *Journal of Insect Conservation*, *20*(2), 165–178. https://doi.org/10.1007/s10841-016-9852-3

Brito, J. G., Martins, R. T., Oliveira, V. C., Hamada, N., Nessimian, J. L., Hughes, R. M., … de Paula, F. R. (2018). Biological indicators of diversity in tropical streams: Congruence in the similarity of invertebrate assemblages. *Ecological Indicators*, *85*(October 2017), 85–92. https://doi.org/10.1016/j.ecolind.2017.09.001

Butchart, S. H. M., Walpole, M., Collen, B., Strien, A. Van, Jörn, P. W., Almond, R. E. A., … Lamarque, J. (2018). Global biodiversity: indicators of recent declines. *Science*, *328*(5982), 1164-1168. https://doi.org/DOI: 10.1126/science.1187512

Campioni, M., & Noack, P. (2012). *Rwanda fast forward: the many perspectives that make Rwanda a unique country. In Rwanda Fast Forward. Palgrave Macmillan, London.*

Clausnitzer, V. & Jödicke, R. (2004). Guardians of the watershed. Global status of dragonflies: critical species, threat and conservation. *International Journal of Odonatology*, (7), 111–143.

Clausnitzer, V., Dijkstra, K. D. B., Koch, R., Boudot, J. P., Darwall, W. R. T., Kipping, J., …

Suhling, F. (2012). Focus on African freshwaters: Hotspots of dragonfly diversity and conservation concern. *Frontiers in Ecology and the Environment*, *10*(3), 129–134. https://doi.org/10.1890/110247

Clausnitzer, V., Kalkman, V. J., Ram, M., Collen, B., Baillie, J. E. M., Bedjanič, M., … Wilson, K. (2009). Odonata enter the biodiversity crisis debate: The first global assessment of an insect group. *Biological Conservation*, *142*(8), 1864–1869. https://doi.org/10.1016/j.biocon.2009.03.028

Conrad, C. C., & Hilchey, K. G. (2011). A review of citizen science and community-based environmental monitoring: Issues and opportunities. *Environmental Monitoring and Assessment*, *176*(1–4), 273–291. https://doi.org/10.1007/s10661-010-1582-5

Cunha, D. G. F., Finkler, N. R., Gómez, N., Cochero, J., Donadelli, J. L., Saltarelli, W. A., … Thomas, S. A. (2019). Agriculture influences ammonium and soluble reactive phosphorus retention in South American headwater streams. *Ecohydrology*, (November), 1–14. https://doi.org/10.1002/eco.2184

Dalu, T., & Froneman, W. (2016). Diatom-based water quality monitoring in southern Africa : challenges and future prospects Diatom-based water quality monitoring in southern Africa : challenges and future prospects. *Water SA*, *42*(4), 551–559. https://doi.org/10.4314/wsa.v42i4.05

Danielsen, F., Burgess, N. D., & Balmford, A. (2005). *Monitoring matters: Examining the potential of locally-based approaches*. *Biodiversity and Conservation* (Vol. 14). https://doi.org/10.1007/s10531-005-8375-0

Dawson, N., Martin, A., Camfield, L., Dawson, N., Martin, A., & Camfield, L. (2019). Can agricultural intensification help attain Sustainable Development Goals ? Evidence from Africa and Asia Development Goals ? Evidence from Africa and Asia. *Third World Quarterly*, *40*(5), 926–946. https://doi.org/10.1080/01436597.2019.1568190

Diao, X., Bahiigwa, G., & Pradesha, A. (2014). *The role of agriculture in the fast-growing Rwandan economy: assessing growth alternatives*. *Intl Food Policy Res Ins* (Vol. 1363).

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Naiman, R. J., Knowler, D. J., & Le, C. (2006). Freshwater biodiversity : importance , threats , status and conservation challenges, (2006), 163–182. https://doi.org/10.1017/S1464793105006950

Dufrene, M. & Legendre, P. (1997). Species assemblages and indicator species : the need for a flexible asymmetrical approach, *67*(3), 345–366. https://doi.org/10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2

Dusková, M., & Machácek, J. (2013). The geomorphological changes caused by the artisanal mining in the mining area Kabera – Rwanda. *International Multidisciplinary Scientific GeoConference: SGEM: Surveying Geology & Mining Ecology Management*, *3*(673.), 673–680.

Dutra, S., & Marco, P. De. (2015). Bionomic differences in odonates and their in fl uence on the ef fi ciency of indicator species of environmental quality, *49*, 132–142. https://doi.org/10.1016/j.ecolind.2014.09.016

Egoh, B. N., Ntshotsho, P., Maoela, M. A., Blanchard, R., Ayompe, L. M., & Rahlao, S. (2020). Setting the scene for achievable post-2020 convention on biological diversity targets : A review of the impacts of invasive alien species on ecosystem services in Africa. *Journal of Environmental Management*, *261*(January), 110171.

https://doi.org/10.1016/j.jenvman.2020.110171

Foley, M. M., Martone, R. G., Fox, M. D., Kappel, C. V., Mease, L. A., Erickson, A. L., … Scarborough, C. (2015). Using ecological thresholds to inform resource management: Current options and future possibilities. *Frontiers in Marine Science*, *2*(NOV), 1–12. https://doi.org/10.3389/fmars.2015.00095

Garcia, X. F., Schnauder, I., Pusch, M. T. (2012). Complex hydromorphology of meanders can support benthic invertebrate diversity in rivers. *Hydrobiologia*, 49–68. https://doi.org/10.1007/s10750-011-0905-z

Gerson Araujo, F., Fichberg, I., Teixeira Pinto, B. C., & Galvao Peixoto, M. (2003). A Preliminary Index of Biotic Integrity for Monitoring the Condition of the Rio Paraiba do Sul, Southeast Brazil. *Environmental Management*. https://doi.org/10.1007/s00267-003-3003-9

Golfieri, B., Hardersen, S., Maiolini, B., & Surian, N. (2016). Odonates as indicators of the ecological integrity of the river corridor: Development and application of the Odonate River Index (ORI) in northern Italy. *Ecological Indicators*, *61*, 234–247. https://doi.org/10.1016/j.ecolind.2015.09.022

Holland, R. A., Darwall, W. R. T., & Smith, K. G. (2012). Conservation priorities for freshwater biodiversity : The Key Biodiversity Area approach refined and tested for continental Africa. *Biological Conservation*, *148*(1), 167–179. https://doi.org/10.1016/j.biocon.2012.01.016

Jeanmougin, M., Leprieur, F., Loïs, G., & Clergeau, P. (2014). Fine-scale urbanization affects Odonata species diversity in ponds of a megacity (Paris, France). *Acta Oecologica*, *59*.

https://doi.org/10.1016/j.actao.2014.05.008

Kalkman, V. J., Boudot, J. P., Bernard, R., De Knijf, G., Suhling, F., & Termaat, T. (2018). Diversity and conservation of European dragonflies and damselflies (Odonata). *Hydrobiologia*, *811*(1), 269–282. https://doi.org/10.1007/s10750-017-3495-6

Kati, V., Devillers, P., Dufrêne, M., Legakis, A., Vokou, D., & Lebrun, P. (2004). Hotspots, complementarity or representativeness? Designing optimal small-scale reserves for biodiversity conservation. *Biological Conservation*, *120*(4), 471–480. https://doi.org/10.1016/j.biocon.2004.03.020

Khan, A. S., Yi, H., Zhang, L., Yu, X., Mbanzamihigo, E., & Umuhumuza, G. (2019). An integrated social-ecological assessment of ecosystem service benefits in the Kagera River Basin in Eastern Africa, 39–53.

Kiffer, W. P., & Marcelo, J. (2017). Structural and functional composition of invertebrate communities associated with leaf patches in forest streams : a comparison between mesohabitats and catchments, 115–127. https://doi.org/10.1007/s10750-017-3249-5

Kindt, R., Breugel, P. V., Lillesø, J. P. B., Bingham, M., Demissew, S., Dudley, C., … Graudal, L. (2011). *Potential Natural Vegetation of Eastern Africa (Ethiopia , Kenya , Malawi , Rwanda , Tanzania , Uganda and Zambia) VOLUME 5 Description and Tree Species Composition for Other Potential Natural Vegetation Types (Vegetation Types other than Forests , Wood* (Vol. 5).

Le Gall, M., Fournier, M., Chaput-Bardy, A., & Husté, A. (2018). Determinant landscape-scale factors on pond odonate assemblages. *Freshwater Biology*, *63*(3), 306–317. https://doi.org/10.1111/fwb.13065

Lowe-McConnell, R. (2010). The changing ecosystem of Lake Victoria, East Africa. *In Freshwater Forum*, *4*(2).

Mangadze, T., Dalu, T., & William Froneman, P. (2019). Biological monitoring in southern Africa: A review of the current status, challenges and future prospects. *Science of the Total Environment*, *648*, 1492–1499. https://doi.org/10.1016/j.scitotenv.2018.08.252

Markert, B. (2007). Definitions and principles for bioindication and biomonitoring of trace metals in the environment, *21*, 77–82. https://doi.org/10.1016/j.jtemb.2007.09.015

Markovic, D., Carrizo, S., Freyhof, J., Cid, N., Lengyel, S., Scholz, M., … Darwall, W. (2014). Europe's freshwater biodiversity under climate change: Distribution shifts and conservation needs. *Diversity and Distributions*, *20*(9), 1097–1107. https://doi.org/10.1111/ddi.12232

Marques, M., Eduardo, P., Samuel, P., & Göran, R. (2018). Predicting the effects of future climate change on the distribution of an endemic damselfly (Odonata , Coenagrionidae) in subtropical South American grasslands, 303–319. https://doi.org/10.1007/s10841-018-0063-y

Martens, K. (2010). The International Year of Biodiversity, 1–2. https://doi.org/10.1007/s10750-009-0045-x

McPeek, M. A. (2008). Ecological factors influencing the distribution and abundances of Odonata. *Dragonflies and Damselfiles: Model Organisms for Ecological and Evolutionary Research*, 51–62. Retrieved from https://antioch.on.worldcat.org/oclc/5104977406

Mendes, T. P., Oliveira-Junior, J. M. B., Cabette, H. S. R., Batista, J. D., & Juen, L. (2017). Congruence and the Biomonitoring of Aquatic Ecosystems: Are Odonate Larvae or Adults the Most Effective for the Evaluation of Impacts. *Neotropical Entomology*, *46*(6), 631–641. https://doi.org/10.1007/s13744-017-0503-5

Miguel, T. B., Oliveira-Junior, J. M. B., Ligeiro, R., & Juen, L. (2017). Odonata (Insecta) as a tool for the biomonitoring of environmental quality. *Ecological Indicators*, *81*(June), 555–566. https://doi.org/10.1016/j.ecolind.2017.06.010

Monteiro-Júnior, C. S., Juen, L., & Hamada, N. (2014). Effects of urbanization on stream habitats and associated adult dragonfly and damselfly communities in central Brazilian Amazonia. *Landscape and Urban Planning*, *127*(1), 28–40. https://doi.org/10.1016/j.landurbplan.2014.03.006

Mu, F., Hoffmann-kroll, R., & Wiggering, H. (2000). Indicating ecosystem integrity — theoretical concepts and environmental requirements, *130*, 13–23. https://doi.org/https://doi.org/10.1016/S0304-3800(00)00210-6

Overdevest, C., Orr, C. H., & Stepenuck, K. (2004). Volunteer stream monitoring and local participation in natural resource issues. *Human Ecology Review*, *11*(2), 177–185. https://doi.org/10.1111/fme.12001

Parmar, T. K., Rawtani, D., & Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. *Frontiers in Life Science*, *9*(2), 110–118. https://doi.org/10.1080/21553769.2016.1162753

Perry, G. L. W., Groeneveld, J., & Miller, B. P. (2010). Development, application and place of neutral theory in ecology and biogeography. *Geography Compass*, *4*(8), 904–918.

https://doi.org/10.1111/j.1749-8198.2010.00354.x

Remsburg, A. J., & Turner, M. G. (2009). Aquatic and terrestrial drivers of dragonfly (Odonata) assemblages within and among north-temperate lakes. *Journal of the North American Benthological Society*, *28*(1), 44–56. https://doi.org/10.1899/08-004.1

Roback, S. S., & Richardson, J. W. (1969). The effects of acid mine drainage on aquatic insects. *Proceedings of the Academy of Natural Sciences of Philadelphia,* 81-107. Retrieved from https://www.jstor.org/stable/4064642?seq=1

Rodrigues, M. E., de Oliveira Roque, F., Quintero, J. M. O., de Castro Pena, J. C., de Sousa, D. C., & De Marco Junior, P. (2016). Nonlinear responses in damselfly community along a gradient of habitat loss in a savanna landscape. *Biological Conservation*, *194*. https://doi.org/10.1016/j.biocon.2015.12.001

Salmah, M. R. C., Tribuana, S. W., & Hassan, A. A. (2006). The population of Odonata (dragonflies) in small tropical rivers with reference to asynchronous growth patterns. *Aquatic Insects*, *28*(3), 195–209. https://doi.org/10.1080/01650420600922315

Samways, M, J., & Simaika, J, P. . (2014). Manual of Freshwater Assessment for South Africa: Dradonfly Biotic Index. *Suricata 2*, *1*. https://doi.org/10.1017/CBO9781107415324.004

Siddig, A. A. H., Ellison, A. M., Ochs, A., Villar-Leeman, C., & Lau, M. K. (2016). How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. *Ecological Indicators*. https://doi.org/10.1016/j.ecolind.2015.06.036

Simaika, J. P., & Samways, M. J. (2009a). An easy-to-use index of ecological integrity for prioritizing freshwater sites and for assessing habitat quality. *Biodiversity and Conservation*, *18*(5), 1171–1185. https://doi.org/10.1007/s10531-008-9484-3

Simaika, J. P., & Samways, M. J. (2009b). Reserve selection using Red Listed taxa in three global biodiversity hotspots: Dragonflies in South Africa. *Biological Conservation*, *142*(3), 638–651. https://doi.org/10.1016/j.biocon.2008.11.012

Simaika, J. P., & Samways, M. J. (2011). Comparative assessment of indices of freshwater habitat conditions using different invertebrate taxon sets. *Ecological Indicators*, *11*(2), 370–378. https://doi.org/10.1016/j.ecolind.2010.06.005

Steubing, L. (1982). Problems of bioindication and the necessity of standardization. Tasks for vegetation science., *7*, 19–24. Retrieved from http://agris.fao.org/agris-search/search.do?recordID=US201302579334

Suganuma, M. S., & Durigan, G. (2015). Indicators of restoration success in riparian tropical forests using multiple reference ecosystems, *23*(3), 238–251. https://doi.org/10.1111/rec.12168

Taniwaki, R. H., Piggott, J. J., Ferraz, S. F. B., & Matthaei, C. D. (2017). Climate change and multiple stressors in small tropical streams. *Hydrobiologia*, *793*(1), 41–53. https://doi.org/10.1007/s10750-016-2907-3

Teder, T., Moora, M., Roosaluste, E., Zobel, K., Pärtel, M., Kõljalg, U., & Zobel, M. (2007). Monitoring of biological diversity: A common-ground approach. *Conservation Biology*, *21*(2), 313–317. https://doi.org/10.1111/j.1523-1739.2006.00575.x

Theodoropoulos, C., Karaouzas, I., Vourka, A., & Skoulikidis, N. (2020). ELF – A benthic macroinvertebrate multi-metric index for the assessment and classification of hydrological alteration in rivers. *Ecological Indicators*, *108*(June 2019), 105713. https://doi.org/10.1016/j.ecolind.2019.105713

Toft, C. A., & Schoener, T. W. (1983). Abundance and Diversity of orb Spiders on 106 Bahamian Islands: Biogeography at an Intermediate Trophic Level. *Island Ecology*, *43*(3). https://doi.org/10.2307/3544101

Turak, E., Harrison, I., Dudgeon, D., Abell, R., Bush, A., Darwall, W., … De Wever, A. (2017). Essential Biodiversity Variables for measuring change in global freshwater biodiversity. *Biological Conservation*, *213*, 272–279. https://doi.org/10.1016/j.biocon.2016.09.005

Uwimana, A., van Dam, A. A., Gettel, G. M., & Irvine, K. (2018). Effects of agricultural land use on sediment and nutrient retention in valley-bottom wetlands of Migina catchment, southern Rwanda. *Journal of Environmental Management*, *219*, 103–114. https://doi.org/10.1016/j.jenvman.2018.04.094

Valente-Neto, F., De Oliveira Roque, F., Rodrigues, M. E., Juen, L., & Swan, C. M. (2016). Toward a practical use of Neotropical odonates as bioindicators: Testing congruence across taxonomic resolution and life stages. *Ecological Indicators*, *61*, 952–959. https://doi.org/10.1016/j.ecolind.2015.10.052

Vorster, C., Samways, M. J., Simaika, J. P., Kipping, J., Clausnitzer, V., Suhling, F., & Dijkstra, K. B. (2020). Development of a new continental-scale index for freshwater assessment based on dragonfly assemblages. *Ecological Indicators*, *109*(April 2019), 105819. https://doi.org/10.1016/j.ecolind.2019.105819

Wyss, K. (2006). *A Thousand Hills for 9 Million People: Land Reform in Rwanda; Restoration of Feudal Order or Genuine Transformation?: FAST Country Risk Profile Rwanda.* Swisspeace. Retrieved from http://edoc.vifapol.de/opus/volltexte/2011/2437/

Zonneveld, I. S. (1983). Principles of bio-indication. In Ecological Indicators for the Assessment of the Quality of Air, Water, Soil, and Ecosystems. In *Springer, Dordrecht.* (pp. 207–217). Retrieved from https://link.springer.com/chapter/10.1007/978-94-009-6322-1_2

Chapter 2: Linking Policy to Practice through Integrated and Adaptive Management of Wetlands in Rwanda

Abstract

Biological diversity and ecological functioning of wetlands has been continuously declining worldwide due to anthropogenic threats. Developing African countries are no exception. It is particularly a daunting challenge to address these threats in densely populated countries, such as Rwanda. To meet tremendous demand for subsistence and national economy, policies often promote practices that adversely affect the environment. Integrated Watershed Management (IWM) and Adaptive Management (AM) conceptual frameworks provide an underpinning holistic platform from which to evaluate the performance of policies and actions on the ground in relation to wetland management. I examine the extent to which environmental policies and practices embrace IWM and AM frameworks in Rwanda, by examining governmental documents for key principles of each framework, particularly in regards to wetlands. Wetlands in Rwanda are particularly vulnerable, given the country has rapidly growing economy and high pollution density. The policy analysis is based on dismembering IWM and AM into their principles. The results show that monitoring and evaluation, a principle of AM is the most commonly included in management, while consideration of multidisciplinarity, one of the IWM principles, is the least. Given the existing political will for AM, I recommend a pragmatic ecological monitoring that can be used for freshwater habitat. This practice can be established with potential to serve and be supported by Citizen Based Monitoring (CBM). CBM could hence be utilized as a platform to instill pro-environmental attitudes within local communities and to set the stage for fostering collaboration between stakeholders, as highlighted by IWM and AM, the underlying conceptual framework of this chapter

Key words: Policy, law, ecological monitoring, wetland, freshwater, integrated management, adaptive management, monitoring.

Introduction

Biological diversity and ecological functioning of wetlands have been continuously declining over the past five decades worldwide due to anthropogenic threats (Clausen & York, 2008; Harvey et al., 2020; Olson et al., 2016; Strayer & Dudgeon, 2010). Developing tropical African countries are no exception (Carrasco et al., 2017; Cobbinah et al., 2015). It is particularly a daunting challenge to address these threats in areas with high population densities, such as Rwanda, one of the most densely populated countries on the African continent (Cobbinah et al., 2015, Jayne et al., 2019).

Heavily human-dominated landscapes such as those found in Rwanda put pressure on ecosystems to meet the tremendous food demand (Muttarak, 2017; Schuyt, 2005; Sievers et al., 2018). Intensified agriculture on hillsides and within wetlands is the primary driver of the loss and degradation of these ecosystems. The situation is exacerbated by policies that promote market-oriented agriculture intended to address the national economic mandate as a major backbone of the country's economy, but that do not take into account the impacts on the environment (Dawson et al., 2019; Nsengimana et al., 2017; Muttarak, 2017). Over the past twenty years, restoration efforts have been growing in response to the degradation of Rwandan ecosystems (Chirwa et al., 2015; McNulty et al., 2016). However, restoration initiatives, have largely focused on habitats that can help rebuild the country's economic capital (e.g. areas of high touristic attraction and hydropower generation (Nabahungu, 2012; Oestigaard, 2010). It not until the last couple of years that wetlands, especially those in urban areas, started gaining attention as awareness about their fragility and ecological importance increases (Nduwayezu et al., 2016).

There is a need to gauge ecological responses to various political and pragmatic interventions on ecosystems. However, the lack of consistent and reliable data on the status of natural ecosystems has been identified as a limiting factor when assessing the link between policy and ecological outcomes (Johns & Eyzaguirre, 2006). Moreover, disciplinary comprehensive and integrated research are lacking as action-response studies have ignored disciplinary and sectorial interdependence (Dalu & Froneman, 2016; Rozzi et al., 2012). These holistic evaluations are invaluable when tracing feedback loops linking management decisions and practices to the status of natural ecosystem. The holistic evaluation could inform not only decision making, but allow for a more comprehensive understanding of the motive behind political decisions from social and economic influences.

This chapter is grounded in the notion that wetlands are integral to our watersheds, an important component of the landscape, and play an instrumental role at the political, socioeconomic and ecological interface. This study stems from a hybrid of two frameworks, Integrated Watershed Management (IWM) and Adaptive Management (AM). As suggested by previous studies (Overdevest et al. 2004, Wang et al. 2016, Wortley et al. 2013), these frameworks provide an underpinning holistic platform from which to evaluate the performance of policies and actions on the ground in relation to wetland management. The adaptive nature of these frameworks stresses the need for ecological monitoring and emphasizes that monitoring can help pinpoint the impacts of specific management practice on the environment, and can be particularly valuable for wetlands management.

This chapter aims at breaking down the walls between disciplines, encouraging policy to consider wetland ecological services and functioning, and to further support long-term monitoring of ecological responses to contemporary wetland management practices in Rwanda, in the face of its fast-growing economy. This chapter begins with a comprehensive overview exposing the shortcomings of wetland conservation efforts in Rwanda through a holistic lens that takes into account an array of elements ranging from ecological to socio-economic and political aspects of wetlands management. To understand political implications to ecosystems, I present a policy and law analysis focusing on IWM and AM elements.

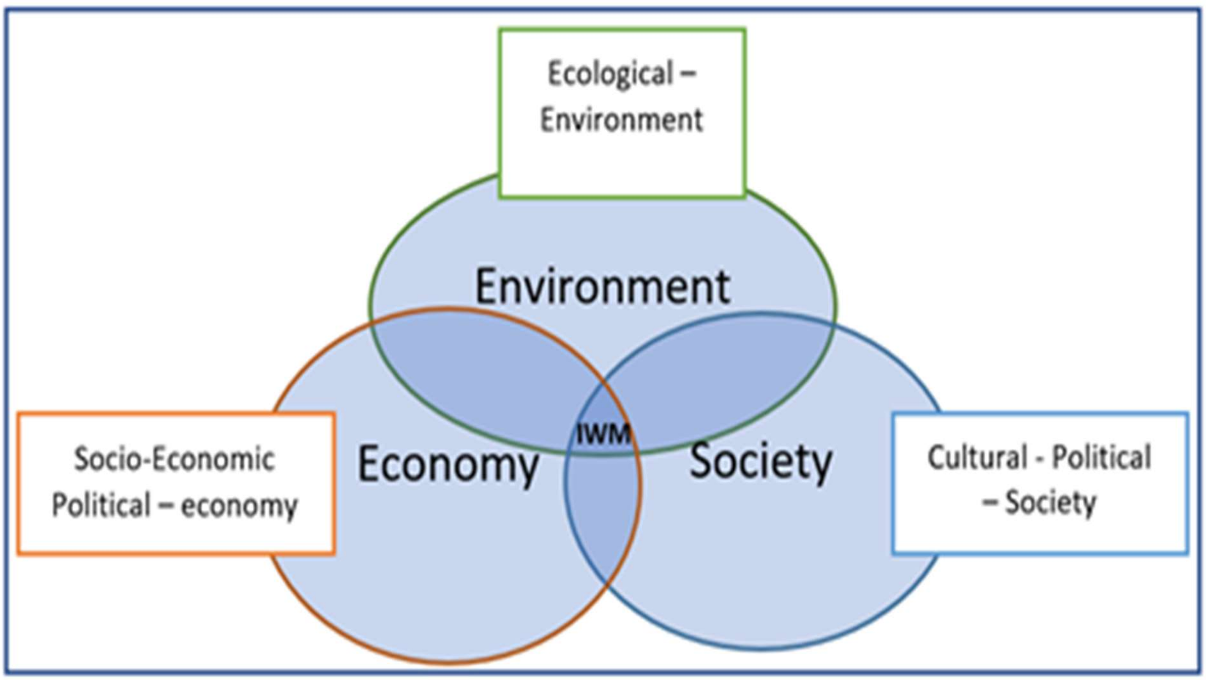

Figure 1-2. Concept of Integrated Watershed Management

Integrated Watershed Management (IWM) encourages the management of watershed components not only as part of natural systems, but also as an interface between nature and humans. This framework includes ecological, cultural, socio-economic and political elements of watershed in their management (Engle et al., 2011; Wang et al., 2016). IWM is a comprehensive and inclusive management approach that takes into account multiple users within a watershed (all actors and sectors) as a way to ensure sustainability of watershed elements, including wetlands. By considering political, economic, and environmental insights (Figure 1-2), IWM strives for a balance between human and environmental needs (Campbell, 2016; Horne et al., 2017).

Some of the major elements of IWM are often joined to strengthen the outcome, depending on the goal of management. Between the ecological, political and economic disciplines, economics is often more practical within the context of environmental management. For example, the combination of social science with economics to form social-economy, is geared toward increasing communication between a multitude of stakeholders and fostering interest and meaning for stakeholders in the management of their wetlands (Blomquist et al., 2005). On the other hand, the combination of politics with economics seeks to decentralize institutions and community groups around management of wetlands, and ensures inclusive participation from the planning stage to implementation of resources management (Engle et al., 2011).

While the interdisciplinary nature of IWM is often better at addressing environmental issues in comparison to conventional management, it is, arguably, a much more complicated method (De Grenade et al., 2016). Conventional management is often more simplistic, and assumes that ecological and socioeconomic elements are consistently predictable over time and space (Moberg & Galaz, 2005). However, more vigilant and advanced perspectives view ecosystems, such as wetlands, as dynamic systems that vary in time and in space (Horne et al., 2017). This complexity further increases when natural ecosystem dynamism is combined with social needs. Therefore, it is advisable to adopt an adaptive style that is flexible as natural changes occur, and socio-economic and political systems evolve. This is precisely where the Adaptive Management concept comes in handy as a supplement to IWM.

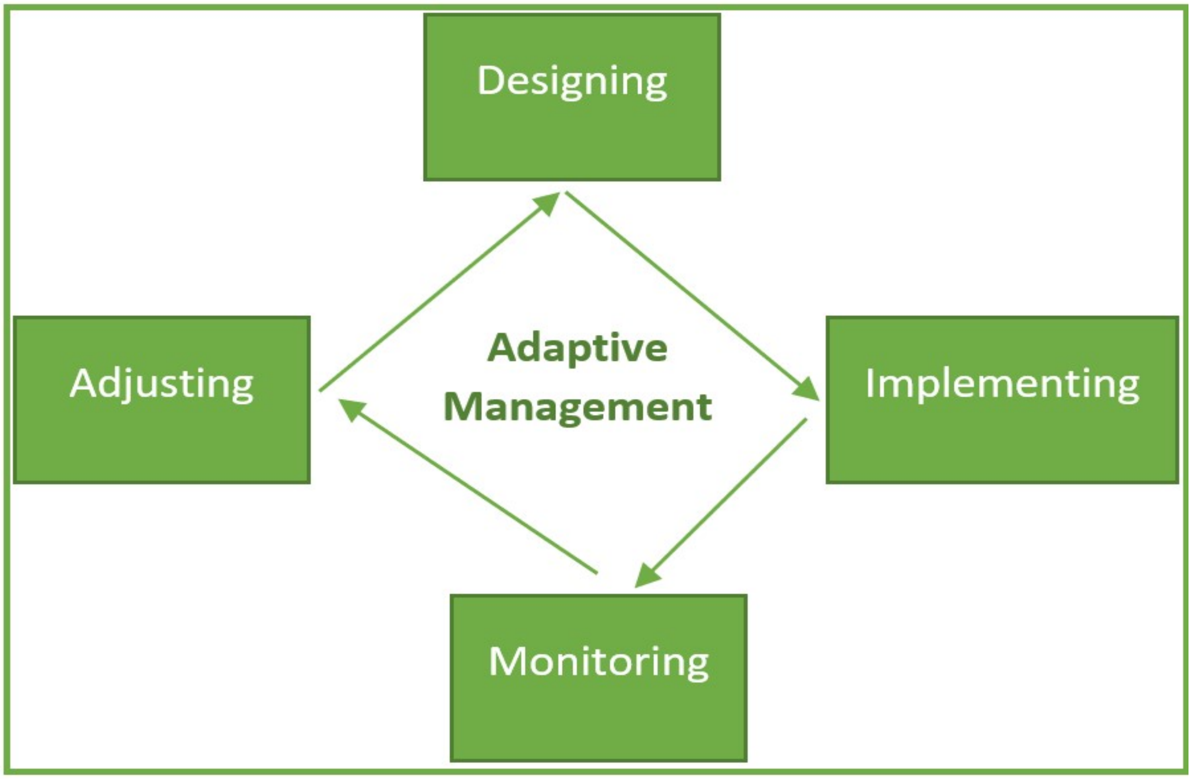

Figure 2-2. Concept of adaptive management

Adaptive Management (AM) was defined by Cosens et al. (2018) as a systematic process for continually improving management strategies while taking into account different alternative values. Similarly, Plummer and Armitage (2013) explain AM as a process that is structured to always consider interventions and policies as experiments (Figure 2-2). The logic here is to keep monitoring feedbacks from implemented actions and accordingly make necessary adjustments based on new insights and experiences learned from past practices. By doing so, AM addresses management mistakes, and the inherent limitation in predicting and controlling drivers of ecosystem change. Also, AM addresses changing ecosystems as they respond to environmental, social economy, political and pragmatic changes (Swyngedouw, 2009).

Numerous studies have suggested that it is practically beneficial to join the IWM and AM conceptual frameworks together in order to address environmental issues, rather than employing each solo (Ozturk et al. 2013, Plummer & Armitage, 2013, Wang et al. 2016). AM has been found beneficial in watershed management due to its primary focus on addressing uncertainties resulting from watershed complexity and changes that IWM bears, although AM is often ineffective alone (Wilhere, 2002). To be truly effective for habitat management, AM should include ecological monitoring data, as well as be informed by all spheres involved, ecological, political and social economic systems (Swyngedouw, 2009).

An overview of wetlands in Rwanda

To discuss the current link between political and pragmatic interventions in wetlands as well as their shortcomings, I present below an overview of wetlands from socio-economic and ecological standpoints.

Socioeconomic aspects

While Rwanda is known for its fast-growing economy, leveraging natural resources has led to alarming environmental degradation. For example, agriculture contributes to nearly 41% of GDP and constitutes over 70% of all exports. However, agriculture stands out as the most threatening factor to the environment, especially wetlands (Nsengimana et al., 2017). Advanced by short-term benefits, agricultural intensification within wetlands is growing as a means to address food and water shortages (Kathiresan, 2011). Such mismanagement of wetlands does not fully support achievement of sustainable development of the nation, which are grounded on green economy(Dawson et al., 2019).

As directed by United Nations' Sustainable Development Goals (SDG) in the Agenda 2030, nations should achieve progress not only in economy, but also in social and environmental dimensions, since these hold each other (United Nations, 2015). The reasoning here is that if the development is to be sustainable, ecosystems need to be used wisely to ensure their goods and services are protected for nations' economy, and support and local human wellbeing. According to Arrow et al. (1995), Cumming et al. (2018), and Dasgupta et al. (2000), development activities that are detrimental to ecological functions often lead to drastic decline in steady supply of goods and services, especially once this impairment reaches a certain threshold. This in turn negatively affects the economy and hampers development.

An example of such a negative feedback can be taken from Rugezi wetland in Rwanda. While the current management of Rugezi wetland is seen as a model of both ecological restoration and local community engagement such as employing local rangers in Rwanda (CEPF, 2018), this wetland once experienced a dramatic water supply shortage due to intensified agriculture and irrigation. This directly affected hydropower generation (Nabahungu, 2012, Sylvère et al., 2016). These shortcomings could be consequences of economic greed and lack of inclusive consultation in the political agenda. As evidence of this, during the initial decision making process for Rugezi intensification, wetland scientists were not brought to the table for advice and local communities' opinions were not considered (Dawson et al., 2019).

As Dawson et al. (2019) argued, a paradigm shift is needed from the use of wetlands for the maximum, short term agricultural productivity to modest, sustainable harvests that account for social economy and cultural values. Agricultural intensification does not give room for poor and small farmers to strive for subsistence (Lambin & Meyfroidt, 2010). Considering only wetland services' extrinsic values (i.e. materialistic values) as the main motives driving wetland preservation is not an effective management plan in the long term (Bland, 2018; Greffiths, 2017).

Additionally, I argue that both agricultural intensification and materialistic-based management fall short since they inhibit local communities from exercising and nurturing sustainable practices passed down through generations. Examples of such practices include the special timing of crop rotation and weeding, elevating plots, terracing, and farming at small scales with diverse crops. These are intended to control pests and soil runoff, as well as maintain soil fertility (Abate et al., 2000; Altieri, 2004). When these practices are not used, the risk of losing this knowledge runs high. This disintegration of sustainable practices may further hinder long-term participation and engagement, which results in shortcomings of long-term wetland management. Over an approximate 30-year period, a number of studies have argued that environmental management must acknowledge the importance of traditional practices in the agricultural sector, as these have sustained habitats prior to the introduction of market-oriented techniques (Sillitoe, 1998; Martin, 2011; Clark, 2005; Wekundah, 2012).

Ecological aspects

It is important to better understand the diversity of wetland ecosystems so as to effectively and appropriately set suitable management plans. Marshlands and swamps in Rwanda are distributed across various ecological zones. These ecological zones are distinguishable based on differences in average precipitation and elevation. Apart from these differences in ecological zones, wetlands can also be grouped into two larger categories. Category one consists of alluvial plains, also known as floodplains, which are those that lie along rivers or adjacent to lakes. Those include, for example, the alluvial plain along Nyabarongo River, Akanyaru River, Akagera Rivers, Lake Kivu, Lake Ihema, Lake Muhazi, and Lake Mugesera. Category two are inland upstream wetlands such as Rugezi and Kamiranzovu (Beuel et al., 2016; Leemhuis et al., 2016). From the hydrological point of view, floodplain wetlands, those located along a riverbed, have the key role to absorb river overflow and as a result control flooding, as opposed to inland wetlands that are located upstream of bigger rivers and serve as storage for ground water and regulator of downstream discharge. Given differences between the two wetland categories, in terms of ecological functions and services, management styles and conservation priorities may also differ depending on specified socio-economic benefits and political agenda.

In addition to wetlands management targeted to wetland functions, there is need to place wetlands into the context of their landscape and watershed. This can provide space for more integrative and collaborative practices. Managing wetlands as components of a larger landscape fosters recognition of interactions residing between and within terrestrial and aquatic ecosystems as well as valley and upslope habitats, both natural and human made from upstream to downstream sections of the watershed.

The landscape-based management of wetlands is important since wetlands are not only affected by on site activities, but also upstream and hillside practices (Uwimana et al., 2018; Weigandt et al., 2015). Wetland management should account for factors that influence runoff and flooding such as rainfall, and the elevation and gradient of the land bordering a wetland (Garcia et al., 2012; Sievers et al., 2018). Moreover, on site flooding is determined by the size of watersheds connected to the wetland and the precipitation rate in the area (Dalzell et al., 2005; Mertes & Warrick, 2001). Therefore, wetlands located in the western part of Rwanda, a region of rolling hills and steep slopes, are prone to high erosion and need special management measures as compared to those in the eastern plains.

Furthermore, wetlands effectiveness in treating loads and filtering water should be considered in setting management plans. It has been found that wetlands must be of sufficient size to allow adequate residence time to treat the loads they receive (Çakir et al., 2015). However, because of the dense hills and broken topography with steep slopes, most wetlands in Rwanda are small in size which makes the accumulation rate of loads happen faster than in other, relatively flatter regions. Finally, given the fact that wetlands store landscape information (through the received loads from hillsides) and can thus reflect the impact of practices on habitats beyond wetlands, ecological monitoring of wetlands can be an efficient way to understand what is going on in the surrounding landscape in terms of ecological degradation (Sievers et al., 2018).

Ecological monitoring

Here, I present ecological monitoring as a core adaptive and collaborative tool of the IWM and AM approach.

***An adaptive process*.** Ecological monitoring is integral to inform conservation planning in understanding the changing state of a habitat in order to allow for responsive management (Engle et al., 2011; Schmeller et al., 2011; Overdevest et al., 2004; Wang et al., 2016). The sustainable use of ecosystems for human use, particularly the most fragile and vulnerable, such as wetlands requires a continuous monitoring of species and ecosystem functions, mainly through biological indication (Danielsen et al., 2005). Species based monitoring programs that use early warning indicators are essential for not only adaptive management, but also for foresight (Burthe et al., 2016).

Environmental stakeholders and policy makers need to recognize biodiversity as an essential and vital element worth integrating into monitoring systems as part of adaptive management processes and ecosystem sustainability (Aavik & Helm, 2018; Falkenmark, 2004). The natural setting of water systems, such as ecology and hydrology offers an opportunity for policy makers and land managers to recognize its interdependence with socioeconomics. Ecological monitoring constitutes a backbone for this holistic and adaptive management of wetlands as it underpins the fundamental elements on which all aspects are grounded (Haase et al., 2018; Török & Helm, 2017).

A collaborative tool. In line with IWM framework, collaboration between sectors and among local communities, ecological experts and decision makers should be at the center of monitoring processes. The monitoring process necessitates insights from not only powerful elites and professional scientists, but also from local lay communities (Danielsen et al., 2005; Aswani et al., 2015). Communities should authentically participate throughout the whole management process, from planning to assessment to formulation of goals. This is supported by the concept of Citizen Based Monitoring (CBM), also known as citizen science, which in its genuine sense, is geared to involving citizens and stakeholders in the management and monitoring of ecosystems (Keough and Blahna, 2006). This is also defined as action that enlists the public in collecting a large amount of ecological or environmental data over a long span of time (Overdevest et al., 2004). It is worth noting that an experimental phase of such a practice of CBM is underway in Rugezi wetland, where local citizens have been engaged in protecting and monitoring the population of grey crowned cranes (CEPF, 2018; Nsengimana et al. 2017).

CBM for freshwater habitats, particularly wetlands, is increasing around the world due to a logistic and educational benefits. Apart from taking less effort and time, the CBM increases awareness and knowledge, which elevates support and advocates for environmentally friendly practices (Keough and Blahna, 2006; Luo et al., 2016). Rwanda presents a particularly conducive environment for CBM through its culturally cohesive society. Rwandan history stresses that Rwandans have cooperative cultural values. For example, members of the community would call upon their family, friends and neighbors to help complete their work (Uwimbabazi & Lawrence, 2013). While the CBM is not only socially suitable but also a financially viable approach for countries like Rwanda, its uptake in East Africa has been very slow (Pocock et al., 2019). CBM is an opportunity to leverage the abundant lay communities and ensure steady habitat monitoring (Lakshminarayanan, 2007). Also, it has been argued that the involvement of amateur citizens, overseen by trained naturalists, can be an answer to the insufficient well-trained workforce. It is relevant to analyze policies and laws in Rwanda relative to what we know about the value of adaptive management and IWM for wetlands management.

Analysis of policies and laws relevant to wetlands management in Rwanda

The present study uses insights from IWM and AM frameworks to analyze how policies in Rwanda are conducive to socially suitable, economically-oriented and ecologically sustainable practices. Rwanda has a number of policies, laws and strategic plans relevant to IWM and AM implementation, which are produced and administered by different governmental policy and regulatory institutions.

I conducted a review of 11 policy-related documents (Table 1-2). These consisted of governmental reports, organic gazettes and various national strategic action plans. For the scope of the selected documents, only governmental reports, policy, law and strategic plan documents published after 2008 were considered, because 2008 was the time significant policy reforms happened as a way to align with the first national Economic Development and Poverty Reduction Strategy, 2008-2013 (EDPRS).

Table 1-2: The reviewed governmental reports, organic gazettes and national strategic action plans.

	Purpose	Focus	Institution	Published year
1	Economy Development and Poverty Reduction Strategy	Economy	Ministry of Finance	2008
2	Guidelines for Environmental Impact Assessment for Wetland Management in Rwanda	Environment, Wetlands	Ministry of Environmental and Natural Resources	2009
3	Mining policy	Mining	Ministry of Forestry and Mining	2010
4	National Disaster Risk Management Plan 2013	Disaster Management	Ministry of Disasters and Refugee affairs	2013
5	Republic of Rwanda (2004). National Land Policy (2004).	Land policy	Ministry of Environmental and Natural Resources	2014
6	National Policy and Strategy for Water Supply and Sanitation Services.	Water and sanitation	Ministry of Infrastructure	2015
7	National Biodiversity Strategy and Action Plan 2016, Water Law 2018	Biodiversity and ecosystems	Ministry of Natural Resources	2016
8	National Sanitation Policy	Sanitation	Ministry of infrastructure	2016
9	National Strategic Plan for Agriculture Transformation	Agriculture	Ministry of Agriculture and	2018
10	Official Gazette no. no.Special of 21/09/2018	National laws	Republic of Rwanda	2018
11	Certification Policy on Suspension	Food Safety	Rwanda Standards Bureau	2019

In each document, I searched for at least one statement that highlight the importance or intention of either of these elements listed in Table 2-2, for a document to be considered as including one of the IWM or AM principles.

Table 2-2: Ranking system used for analysis of laws and policies related to wetland management in Rwanda

Elements of IWM Considered	Element of AM considered
Inclusiveness or vertical and horizontal Consultation/collaboration	Monitoring and/or evaluation
Inter/multi-disciplinary consideration	Flexibility to change/or adapt
One of the above	One of the above
None of the above	None of the above

The analysis focused on identifying explicit plans for inclusion of diverse sectors or/and disciplines in environmental issues, within policies and laws governing freshwater habitats in Rwanda. As delimitation of the analysis, I did not look at the process and efficiency of putting policy and laws into effect, while it is important to take into account possible discrepancy between policy formulation and implementation. This analysis rather envisages revealing: (1) commitments in mainstreaming environmental elements into other sectors, and collaboration between stakeholders, (2) consideration of interdisciplinary nature of environment, (3) recognition of importance of monitoring-based management, (4) flexibility or commitment to adjust based on the learned experience (adaptive approach).

My hypothesis was that all the reviewed documents highlight statements that reflect the intention to consider inclusion, collaboration, and interdisciplinarity as per IWM framework. I also assessed whether these documents included the existence of the AM elements of evaluation and monitoring, as well as flexibility for change as informed by experience and learning. If these hypotheses are true, I assume there is a will to nurture IWM and AM principles at policy formulation level, thus a potential space for their implementation.

Results

Wetland Management in Rwanda through IWM and AM lenses

The policy analysis shows that 30% of the 11 documents analyzed highlight IWM elements including openness to inclusion, consultation or collaboration, while the importance of interdisciplinary and multidisciplinary in management is mentioned in 16.6% of the documents. As for AM, 30% and 23.3% of documents, respectively mention the role of monitoring and flexibility to change from the learned experience (Figure 3-2).

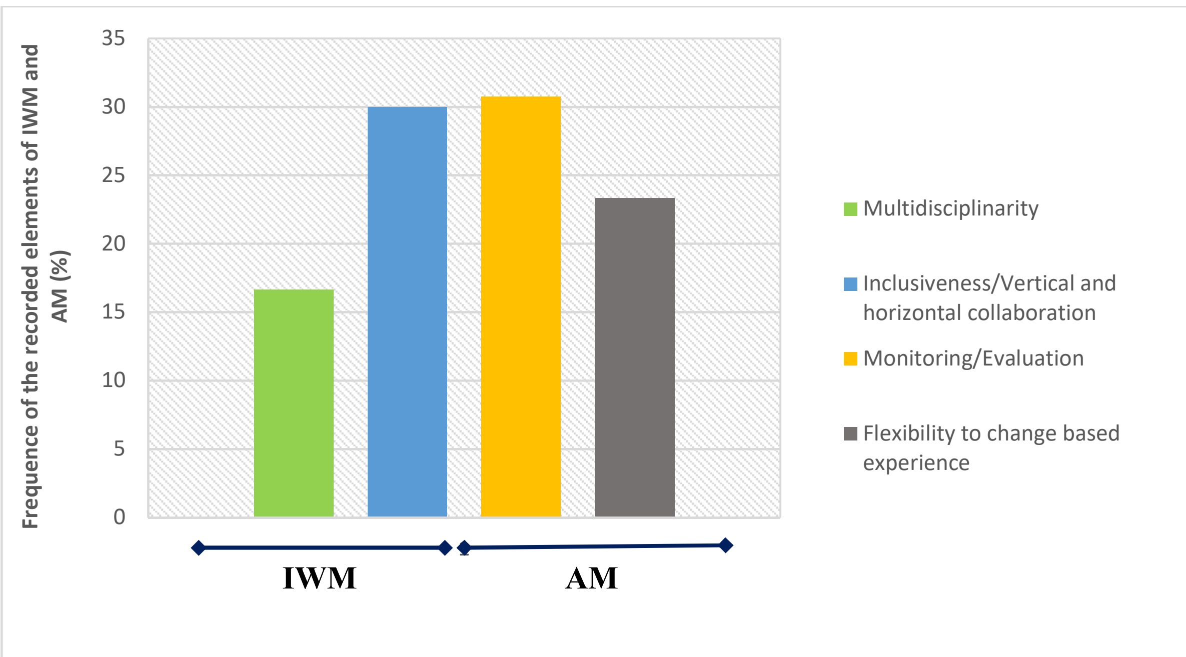

Figure 3-2. Analysis of laws and policies that incorporate principles of AM and IWM framework. This is based on analysis of presence or absence of AM or IWM principles within the 13 policy-related documents.

Discussion

Policies governing wetlands and reflection on integrated and adaptive management

Inclusion can be considered as the key element of the IWM. This element gets its complete meaning when it considers interdisciplinary and multidisciplinary nature of management (Ozturk et al. 2013, Plummer & Armitage, 2013, Wang et al. 2016). In other words, policies that promote inclusion in wetlands management also tend to connect disciplines with potential influence to make the management a success. For example, some of the statements of the national land policy, one of the analyzed policies, encourage integration of social and natural science principles to political and decision-making processes (Republic of Rwanda, 2004). However, other studies have shown that inclusion is not fully achieved until public involvement in decision-making process is added to ecological based-understanding for ecosystem management (Endter-Wada et al., 1998; Khan et al., 2019).

One could speculate that collaboration and consideration of interdisciplinary and multidisciplinary nature should be prominent in wetlands management in Rwanda. The need for this is clearly obvious, looking at the transboundary nature of wetlands. This showcases the importance of collaboration among stakeholders, as compartmentalized within different political boundaries(Lubner, 2015). For example, to face such challenges imposed by the biophysical nature of wetlands, Rwanda chose to adopt several regional policies, most of which are grounded on collaboration and consider interdisciplinary and multidisciplinary nature for environmental management. These include Lake Victoria Environment Management Programme, Nile Basin Initiative (NBI), and Kagera Transboundary Agro-Ecosystems management (Salman, 2013). Such a collaborative will in the Rwandan government is also exemplified by the ratified international treaties and conventions (Republic of Rwanda, 2011).

The international treaties have created spillovers at the national level. The national Biodiversity Strategic Action Plan (NBSAP) in Rwanda has been developed to comply with the multilateral treaty, the Convention on Biological Diversity (CBD). Ratified in 1995, the CBD states that countries have full sovereignty over the ownership of biodiversity and natural resources. The Rwandan NBSAP (2016) recognizes the local biodiversity crises and has commitments to support biodiversity related policies through inclusive principles as per IWM. The NBSAP emphasizes the need for inclusion of biodiversity conservation in economic and development sectors such as agriculture and animal resources, fisheries, forestry, mining and infrastructures. While the NBSAP embraces the IWM and AM concepts, the question remains as to whether other sectors take into account the NBSAP in their strategic plans. I do think that the successful implementation of NBSAP is dependent on the integrative and adaptative nature of other sectors' structures, ecological monitoring could be one of ways to evaluate the NBSAP.

Consideration of ecological monitoring in wetland management

IWM and AM principles can also be supported by landscape-based resources concept as a spatially inclusive framework (Weigandt et al., 2015). A couple of examples show how the Rwandan government recognizes the need to manage ecosystems in spatially integrated manner. As per Article 7 of the Official Gazette Special of 21/09/2018, water resource management should acknowledge the interests of all water users, land and other natural resources. The law highlights the role of these users and their entitlement to participate in water resources planning and management, through representatives. However, this law does not include coordination mechanisms to promote the involvement of multiple stakeholders, as suggested by the IWM framework.

Not only is there limited coordination among water users based on the document analysis, but the ability for legislation to adapt quickly to environmental changes is low (Figure 3-2). Also, the NBSAP (2016), one of the reviewed documents, pointed out a number of drawbacks including (1) lack of coordination of intervention and dialogue among actors (2) absence of decentralized structure for grassroot actions, and (3) deficiency in considering biodiversity and other natural settings in management.

I think the first drawback is meant, in other words, to highlight the lack of inclusion of scientists among other actors. The lack of inclusion can be noticeable in how wetlands are defined in the national legislation. For example, the Rwandan organic law, official gazette of 21/09/2018, defines wetlands as a flat area made up of valleys and plainlands with much stagnant water and biodiversity such as papyrus, cypress or other vegetation of the same family (Republic of Rwanda, 2018). The legislative definition of wetlands does not separate artificial from natural wetlands. Also, it does not categorize permanent versus temporal types of wetlands. It has been shown that the seasonal extremes are growing more and more as result of climate change. The same argument was supported by Nyandwi (2016) who pointed out that there was confusion in the results from wetlands inventories. The inconsistency in the tallying of the number of wetlands in Rwanda is apparent with more wetlands in wet season and less in dry season. This creates confusion among conservation actors while prioritizing sites of high protection concern. A clear definition of wetland habitats based on ecological principles in legislation is a crucial step for conservation and management of these habitats.

The second limitation reflects the gaps between policy entities and local community engagement. As noted by Petts (2007), the limited community engagement currently taking place is seen as the exception rather than the normal process. Policies should be formulated in consultation with local communities (Kin et al., 2016). The decision making should involve the community at all levels. This style of decision-making fosters strong and long-term partnerships, and empowers the community. For Rwanda, as highlighted in Figure 3-1, among the principals of IWM and AM, involvement and consultation appear to be the most common currently used in policies and strategies, but more involvement, especially that which influences decision making, is needed at the local level. In order to make decisions that appeal to local communities, the involvement should be part of each step, cutting across a spectrum of identification of policy needed, inquiry and setting policy that address the issue (Danielsen et al., 2005; Parkes & Panelli, 2001).

The third limitation, raised by the NBSAP (2016), emphasized the need for ecological and biodiversity-based data in management as per AM framework. In addition to the relevance of IWM elements in wetlands management discussed earlier, AM is important to mention here, given the results of this analysis show that a bit more 30% explicitly outline the importance of monitoring and evaluation in ecosystem monitoring. This political endeavor is consistent with earlier studies highlighting the role of AM through ecological monitoring. This is particularly needed given the growing human impacts on ecosystems. Timely and regular ecological monitoring can elevate a better understanding and foresight the non-linear dynamism of ecosystems (Danielsen et al., 2005; Foley et al., 2015). Watershed based monitoring can serve as a vehicle to gathering data needed to inform ecosystem and land managers (Verdone & Seidl, 2016; Renner et al., 2018).

In this context of watershed or landscape-based monitoring, it appears that some of the protected area's boundaries have been set without considering the water resource systems. For example, within one of the catchments of the Congo Nile Crest watershed, the integrity of freshwater catchments in the newly created Gishwati-Mukura National Park was investigated. Reflected in biological indicators, the results show how highly streams are impaired due to a strong impact from outside of the park (Uyizeye et al., in prep.). It was observed that headwaters are found within crop and cattle farms around Gishwat-Mukura National Park. In this regard, a few square kilometers expansion of reforestation and protection would suffice to cover the major headwaters that feed into streams crossing the park (Uyizeye et al., in prep).

Conclusion and recommendations

- Overall, this chapter highlights the exiting will and needs for integration of comprehensive and adaptive approaches for sustainable wetlands management in Rwanda's policies.
- Through an IWM and AM lenses, I point out gaps that lead to limited inclusion of stakeholders and integration of adaptive principles. These gaps can lead to unrealistic planning and establishment of unachievable goals.
- More in-depth studies focusing on the ecological piece of the holistic interdisciplinary field of freshwater ecosystem management are needed. This includes a deeper investigation of the extent to which different land use types, as shaped by political and socio-economic drivers, affect freshwater habitats in Rwanda. I recommend development of sensitive biological indicators with an early warning ability could be appropriate for the unique ecological and social economic Rwandan landscape.

- The biological monitoring indicators provide valuable information for environmental policy decision-making. Also, this practice can be established with potential to serve and be supported by Citizen Based Monitoring (CBM). CBM could hence be utilized as a platform to instill pro-environmental attitudes within local communities and to set the stage for fostering collaboration between stakeholders, as highlighted by IWM and AM, the underlying conceptual framework of this study.

References

Aavik, T., & Helm, A. (2018). Restoration of plant species and genetic diversity depends on landscape-scale dispersal. *Restoration ecology*, *26*, S92-S102. https://doi.org/10.1111/rec.12634

Abate, T., van Huis, A., & Ampofo, J. K. O. (2000). Pest management strategies in traditional agriculture: an African perspective. *Annual review of entomology*, *45*(1), 631-659. https://doi.org/10.1146/annurev.ento.45.1.631

Altieri, M. A. (2004). Linking ecologists and traditional farmers in the search for sustainable agriculture. *Frontiers in Ecology and the Environment*, *2*(1), 35-42. https://doi.org/10.1890/1540-9295(2004)002[0035:LEATFI]2.0.CO;2

Arrow, K., Bolin, B., Costanza, R., Dasgupta, P., Folke, C., Holling, C. S., ... & Pimentel, D. (1995). Economic growth, carrying capacity, and the environment. *Ecological economics*, *15*(2), 91-95. https://doi.org/10.1016/0921-8009(95)00059-3

Aswani, S., Vaccaro, I., Abernethy, K., Albert, S., & de Pablo, J. F. L. (2015). Can perceptions of environmental and climate change in island communities assist in adaptation planning locally? *Environmental management*, *56*(6), 1487-1501. https://doi.org/10.1007/s00267-015-0572-3

Beuel, S., Alvarez, M., Amler, E., Behn, K., Kotze, D., Kreye, C., ... & Becker, M. (2016). A rapid assessment of anthropogenic disturbances in East African wetlands. *Ecological indicators*, *67*, 684-692. https://doi.org/10.1016/j.ecolind.2016.03.034

Bland, E. (2018). Closing the rift between agriculture and conservation: explorations of food sovereignty on an island in the salish sea (Doctoral dissertation). University of Victoria. URI: https://dspace.library.uvic.ca//handle/1828/10479

Blomquist, W., & Schlager, E. (2005). Political pitfalls of integrated watershed management. *Society and Natural Resources*, *18*(2), 101-117. https://doi.org/10.1080/08941920590894435

Burthe, S. J., Henrys, P. A., Mackay, E. B., Spears, B. M., Campbell, R., Carvalho, L., … Daunt, F. (2016). Do early warning indicators consistently predict nonlinear change in long-term ecological data ?, 666–676. https://doi.org/10.1111/1365-2664.12519

Çakir, R., Gidirislioglu, A., & Çebi, U. (2015). A study on the effects of different hydraulic loading rates (HLR) on pollutant removal efficiency of subsurface horizontal-flow constructed wetlands used for treatment of domestic wastewaters. *Journal of environmental management*, *164*, 121-128. https://doi.org/10.1016/j.jenvman.2015.08.037

Campbell, I. C. (2016). Integrated management of large rivers and their basins. *Ecohydrology & Hydrobiology*, *16*(4), 203-214. https://doi.org/10.1016/j.ecohyd.2016.09.006

Carrasco, L. R., Le Nghiem, T. P., Chen, Z., & Barbier, E. B. (2017). Unsustainable development pathways caused by tropical deforestation. *Science advances*, *3*(7), e1602602. https://doi.org/10.1126/sciadv.1602602

CEPF. (2018). Strengthening Rugezi marsh rangers to increase protection of Rugezi Marsh and its iconic Grey Crowned Cranes, *CEPF Grant*, *Rwanda Wildlife Conservation Association*.

Chirwa, P. W., Larwanou, M., Syampungani, S., & Babalola, F. D. (2015). Management and restoration practices in degraded landscapes of Eastern Africa and requirements for up-scaling. *International Forestry Review*, *17*(3), 20-30. https://doi.org/10.1505/146554815816007009

Clark, D. A. T. (2005). Indigenous voice and vision as commodity in a mass-consumption society: The colonial politics of public opinion polling. *American Indian Quarterly*, 228-238. https://www.jstor.org/journal/amerindiquar

Clausen, R. and York, R. (2008). Global biodiversity decline of marine and freshwater fish: a cross-national analysis of economic, demographic, and ecological influences. *Social Science Research*, *37*(4), 1310-1320. https://doi.org/10.1016/j.ssresearch.2007.10.002

Cobbinah, P. B., Erdiaw-Kwasie, M. O., & Amoateng, P. (2015). Rethinking sustainable development within the framework of poverty and urbanisation in developing countries. *Environmental Development*, *13*, 18-32. https://doi.org/10.1016/j.envdev.2014.11.001

Cosens, B. A., Gunderson, L., & Chaffin, B. C. (2018). Introduction to the Special Feature Practicing Panarchy. *Ecology and Society*, *23*(1). https://doi.org/10.5751/ES-09524-230104

Cumming, G. S., & von Cramon-Taubadel, S. (2018). Linking economic growth pathways and environmental sustainability by understanding development as alternate social–ecological regimes. *Proceedings of the National Academy of Sciences*, *115*(38), 9533-9538. https://doi.org/10.1073/pnas.1807026115

Dalu, T., & Froneman, W. (2016). Diatom-based water quality monitoring in southern Africa : challenges and future prospects Diatom-based water quality monitoring in southern Africa : challenges and future prospects. *Water SA*, *42*(4), 551–559. https://doi.org/10.4314/wsa.v42i4.05

Dalzell, B. J., Filley, T. R., & Harbor, J. M. (2005). Flood pulse influences on terrestrial organic matter export from an agricultural watershed. *Journal of Geophysical Research:*

Biogeosciences, *110*(G2). https://doi.org/10.1029/2005JG000043

Danielsen, F., Burgess, N. D., & Balmford, A. (2005). *Monitoring matters: Examining the potential of locally-based approaches*. *Biodiversity and Conservation* (Vol. 14). https://doi.org/10.1007/s10531-005-8375-0

Dasgupta, P., Levin, S., & Lubchenco, J. (2000). Economic pathways to ecological sustainability. *BioScience*, *50*(4), 339-346. https://doi.org/10.1641/0006-3568(2000)050[0339:EPTES]2.3.CO;2

Dawson, N., Martin, A., Camfield, L., Dawson, N., Martin, A., & Camfield, L. (2019). Can agricultural intensification help attain Sustainable Development Goals ? Evidence from Africa and Asia Development Goals ? Evidence from Africa and Asia. *Third World Quarterly*, *40*(5), 926–946. https://doi.org/10.1080/01436597.2019.1568190

De Grenade, R., House-Peters, L., Scott, C. A., Thapa, B., Mills-Novoa, M., Gerlak, A., & Verbist, K. (2016). The nexus: reconsidering environmental security and adaptive capacity. *Current opinion in environmental sustainability*, *21*, 15-21. https://doi.org/10.1016/j.cosust.2016.10.009

Endter-Wada, J., Blahna, D., Krannich, R., & Brunson, M. (1998). A framework for understanding social science contributions to ecosystem management. *Ecological Applications*, *8*(3), 891–904. https://doi.org/10.1890/1051-0761(1998)008[0891:AFFUSS]2.0.CO;2

Engle, N. L., Johns, O. R., Lemos, M. C., & Nelson, D. R. (2011). Integrated and adaptive management of water resources: tensions, legacies, and the next best thing. *Ecology and society*, *16*(1). URL: http://www.ecologyandsociety.org/vol16/iss1/art19/

Falkenmark, M. (2004). Towards integrated catchment management: opening the paradigm locks between hydrology, ecology and policy-making. *International Journal of Water Resources Development*, *20*(3), 275-281. https://doi.org/10.1080/0790062042000248637

Foley, M. M., Martone, R. G., Fox, M. D., Kappel, C. V., Mease, L. A., Erickson, A. L., … Scarborough, C. (2015). Using ecological thresholds to inform resource management: Current options and future possibilities. *Frontiers in Marine Science*, *2*(NOV), 1–12. https://doi.org/10.3389/fmars.2015.00095

Greffiths, I. J. (2017). Influence of governance institutions on households' willingness to pay for resources conservation in Khalong-la-Lithunya wetland area Lesotho (*Doctoral dissertation, University of Pretoria).* URI: http://hdl.handle.net/2263/60806

Haase, P., Tonkin, J. D., Stoll, S., Burkhard, B., Frenzel, M., Geijzendorffer, I. R., ... & Mirtl, M. (2018). The next generation of site-based long-term ecological monitoring: Linking essential biodiversity variables and ecosystem integrity. *Science of the Total Environment*, *613*, 1376-1384. https://doi.org/10.1016/j.scitotenv.2017.08.111

Harvey, J. A., Heinen, R., Armbrecht, I., Basset, Y., Baxter-Gilbert, J. H., Bezemer, T. M., ... & Clausnitzer, V. (2020). International scientists formulate a roadmap for insect conservation and recovery. *Nature Ecology & Evolution*, 1-3.

Horne, A. C., O'Donnell, E. L., Webb, J. A., Stewardson, M. J., Acreman, M., & Richter, B. (2017). The environmental water management cycle. In *Water for the Environment* (pp. 3-16). Academic Press. https://doi.org/10.1016/B978-0-12-803907-6.00001-2

Jayne, T. S., Snapp, S., Place, F., & Sitko, N. (2019). Sustainable agricultural intensification in an era of rural transformation in Africa. *Global food security*, *20*, 105-113. https://doi.org/10.1016/j.gfs.2019.01.008

Johns, T., & Eyzaguirre, P. B. (2006). Linking biodiversity, diet and health in policy and practice. *Proceedings of the Nutrition Society*, *65*(2), 182-189. https://doi.org/10.1079/PNS2006494

Kathiresan, A. (2011). Strategies for sustainable crop intensification in Rwanda. Shifting focus from producing enough to producing surplus. *Ministry of Agriculture and Animal Resources.*http://197.243.22.137/ruhango8/fileadmin/templates/Document/CIP_Strategies_2011.pdf

Keough, H. L., & Blahna, D. J. (2006). Achieving integrative, collaborative ecosystem management. *Conservation Biology*, *20*(5), 1373-1382. https://doi.org/10.1111/j.1523-1739.2006.00445.x

Khan, A. S., Yi, H., Zhang, L., Yu, X., Mbanzamihigo, E., Umuhumuza, G., ... & Yevide, S. I. A. (2019). An integrated social-ecological assessment of ecosystem service benefits in the Kagera River Basin in Eastern Africa. *Regional environmental change*, *19*(1), 39-53. https://www.springer.com/journal/10113

Kin, E., Storey, R., Wright-Stow, A. E., & Davies-Colley, R. J. (2016*).* Engaging communities in freshwater monitoring: benefits and challenges. *Landcare Research.* https://niwa.co.nz/

Lakshminarayanan, S. (2007). Using citizens to do science versus citizens as scientists. *Ecology and Society*, *12*(2). URL: http://www.ecologyandsociety.org/vol12/iss2/resp2/

Lambin, E. F., & Meyfroidt, P. (2010). Land use transitions: Socio-ecological feedback versus socio-economic change. *Land use policy*, *27*(2), 108-118. https://doi.org/10.1016/j.landusepol.2009.09.003

Leemhuis, C., Amler, E., Diekkrüger, B., Gabiri, G., & Näschen, K. (2016). East African wetland-catchment data base for sustainable wetland management. *Proceedings of the*

International Association of Hydrological Sciences, *374*, 123-128. https://doi.org/10.5194/piahs-374-123-2016

Lubner, V. E. (2015). The Impact of International Water Treaties on Transboundary Water Conflicts : A Study Focused on Large Transboundary Lakes, (May).

Luo, L., Qin, L., Wang, Y., & Wang, Q. (2016). Environmentally-friendly agricultural practices and their acceptance by smallholder farmers in China—A case study in Xinxiang County, Henan Province. *Science of the Total Environment*, *571*, 737-743. https://doi.org/10.1016/j.scitotenv.2016.07.045

Martin, D. (2011). Indigenous Methodologies: Characteristics, Conversations, and Contexts. *Canadian Journal of Public Health*, *102*(3), 237. https://www.questia.com/#/library

McNulty, S., Mack, E. C., Sun, G., & Caldwell, P. (2016). Hydrologic modeling for water resource assessment in a developing country: the Rwanda case study. *In: Lachassagne, P. and M. Lafforgue (eds).* Forest and the Water Cycle: Quantity, Quality, Management. Cambridge Scholars Publishing., 2016, 181-203.

Mertes, L. A., & Warrick, J. A. (2001). Measuring flood output from 110 coastal watersheds in California with field measurements and SeaWiFS. *Geology*, *29*(7), 659-662.https://doi.org/10.1130/0091-7613(2001)029<0659:MFOFCW>2.0.CO;2

Moberg, F., & Galaz, V. (2005). Resilience: going from conventional to adaptive freshwater management for human and ecosystem compatibility. *Swedish Water House Policy Brief*, *3*. URI:http://hdl.handle.net/10535/5136

Mukashema, A., Veldkamp, T., & Amer, S. (2016). Si1ty percent of small coffee farms have suitable socio-economic and environmental locations in Rwanda. *Agronomy for sustainable development*, *36*(2), 31. https://doi.org/ 10.1007/s13593-016-0363-0

Muttarak, B. R. (2017). Focusing on demographic differential vulnerability, *Culture, Beliefs and the Environment, 1-8.* https://www.populationenvironmentresearch.org/cyberseminars

Nabahungu, N. L. (2012). Problems and opportunities of wetland management in Rwanda Nsharwasi Léon Nabahungu. *Thesis Wageningen University, the Netherlands (2012)*. ISBN 978-90-8585-924-6. https://library.wur.nl/WebQuery/wurpubs/423369

Nduwayezu, G., Sliuzas, R., & Kuffer, M. (2016). Modeling urban growth in Kigali city Rwanda. *Rwanda Journal*, *1*(1S). DOI: 10.4314/rj.v1i2S.7D

NBSAP (2016). Rwanda National Strategic Plan. *Minister of Natural Resources*

Nsengimana, V., Weihler, S., & Kaplin, B. A. (2017). Perceptions of Local People on the Use of Nyabarongo River Wetland and Its Conservation in Rwanda Perceptions of Local People on the Use of Nyabarongo River. *Society & Natural Resources*, *30*(1), 3–15. https://doi.org/10.1080/08941920.2016.1209605

Nyandwi, E., Veldkamp, T., & Amer, S. (2016). Regional climate sensitivity of wetland environments in Rwanda : the need for a location-specific approach. *Regional Environmental Change*, *16*(6), 1635–1647. https://doi.org/10.1007/s10113-015-0905-z

Oestigaard, T. (2010). Nile issues: Small streams from the Nile Basin Research Programme. *Fountain Publishers.* URI: http://hdl.handle.net/1956/4018

Olson, D. M., Dinerstein, E., Powell, G. V. N., & Eric, D. (2016). Society for Conservation Biology Conservation Biology for the Biodiversity Crisis Conservation Biology for the Biodiversity Crisis, *16*(1), 1–3. https://doi.org/10.1046/j.1523-1739.2002.01612.x

Overdevest, C., Orr, C. H., & Stepenuck, K. (2004). Volunteer stream monitoring and local participation in natural resource issues. *Human Ecology Review*, *11*(2), 177–185. https://doi.org/10.1111/fme.12001

Ozturk, I., Erturk, A., Ekdal, A., Gurel, M., Cokgor, E., Insel, G., ... & Tanik, A. (2013). Integrated watershed management efforts: case study from Melen Watershed experiencing interbasin water transfer. *Water Science and Technology: Water Supply*, *13*(5), 1272-1280. https://doi.org/10.2166/ws.2013.139

Parkes, M., & Panelli, R. (2001). Integrating catchment ecosystems and community health: The value of participatory action research. *Ecosystem Health*, *7*(2), 85–106. https://doi.org/10.1046/j.1526-0992.2001.007002085.x

Petts, J. (2007). Learning about learning: lessons from public engagement and deliberation on urban river restoration. *Geographical journal*, *173*(4), 300-311. https://doi.org/10.1111/j.1475-4959.2007.00254.x

Plummer, R., Armitage, D. R., & De Loë, R. C. (2013). Adaptive comanagement and its relationship to environmental governance. *Ecology and Society*, *18*(1). https://doi.org/10.1016/j.ecolecon.2006.09.025

Pocock, M. J. O., Roy, H. E., August, T., Kuria, A., Barasa, F., Bett, J., … Trevelyan, R. (2019). Developing the global potential of citizen science : Assessing opportunities that benefit people , society and the environment in East Africa, (August 2018), 274–281. https://doi.org/10.1111/1365-2664.13279

Renner, S., Périco, E., Dalzochio, M. S., & Sahlén, G. (2018). Water body type and land cover shape the dragonfly communities (Odonata) in the Pampa biome, Rio Grande do Sul, Brazil. *Journal of insect conservation*, *22*(1), 113-125.

https://www.springer.com/journal/10841

Republic of Rwanda (2004). National Land Policy (2004). https://rlma.rw

Republic of Rwanda (2011). Rwanda Biodiversity Policy (2011). http://rw.chm-cbd.net

Republic of Rwanda (2018). Official Gazette no.Special of 21/09/2018. https://environment.gov.rw

Rozzi, R., Armesto, J. J., Gutiérrez, J. R., Massardo, F., Likens, G. E., Anderson, C. B., ... & Kennedy, J. H. (2012). Integrating ecology and environmental ethics: earth stewardship in the southern end of the Americas. *BioScience*, *62*(3), 226-236. https://doi.org/10.1525/bio.2012.62.3.4

Salman, S. M. (2013). The Nile Basin cooperative framework agreement: A peacefully unfolding African spring? *Water International*, *38*(1), 17-29. https://doi.org/10.1080/02508060.2013.744273

Schmeller, D. S., Julliard, R., Bellingham, P. J., Böhm, M., Brummitt, N., Chiarucci, A., ... & Gregory, R. D. (2015). Towards a global terrestrial species monitoring program. *Journal for Nature Conservation*, *25*, 51-57. https://doi.org/10.1016/j.jnc.2015.03.003

Schuyt, K. D. (2005). Economic consequences of wetland degradation for local populations in Africa. *Ecological economics*, *53*(2), 177-190. https://doi.org/10.1016/j.ecolecon.2004.08.003

Sievers, M., Hale, R., Parris, K. M., & Swearer, S. E. (2018). Impacts of human-induced environmental change in wetlands on aquatic animals. *Biological Reviews*, *93*(1), 529-554. https://doi.org/10.1111/brv.12358

Sillitoe, P. (1998). Knowing the land: soil and land resource evaluation and indigenous knowledge. *Soil use and Management*, *14*(4), 188-193. https://doi.org/10.1111/j.1475-

2743.1998.tb00148.x

Strayer, D. L., & Dudgeon, D. (2010). Freshwater biodiversity conservation: recent progress and future challenges. *Journal of the North American Benthological Society*, *29*(1), 344-358. DOI: 10.1899/08-171.1

Swyngedouw, E. (2009). The political economy and political ecology of the hydro-social cycle. *Journal of Contemporary Water Research & Education*, *142*(1), 56-60. https://doi.org/10.1111/j.1936-704X.2009.00054.x

Sylvère, H., & Emmanuel, T. (2016). Using social network analysis to understand actor participation and influence on sustainable management of Rugezi wetland, Rwanda. *Rwanda Journal*, *1*(1S). http://dx.doi.org/10.4314/rj.v1i1S.12D

Török, P., & Helm, A. (2017). Ecological theory provides strong support for habitat restoration. *Biological Conservation*, *206*, 85–91. https://doi.org/10.1016/j.biocon.2016.12.02.24

United Nations (2015). Transforming Our World: The 2030 Agenda for Sustainable Development. Resolution. http://www.un.org/ga/search/view_doc.asp?symbol=A/RES/70/1&Lang=E

United Nations (2017) World Population Prospects: The 2017 Revision.

Uwimana, A., van Dam, A. A., & Irvine, K. (2018). Effects of conversion of wetlands to rice and fish farming on water quality in valley bottoms of the Migina catchment, southern Rwanda. *Ecological engineering*, *125*, 76-86. https://doi.org/10.1016/j.ecoleng.2018.10.019

Uwimbabazi, P., & Lawrence, R. (2013). Indigenous Practice, Power and Social Control: The Paradox of the Practice of Umuganda in Rwanda. *Race, Power and Indigenous*

Knowledge Systems, 248.
http://utlo.ukzn.ac.za/Files/Alternation%2020.1%20(2013).pdf#page=253

Uyizeye, E., Kaplin A.B., Willey, L., Clausnitzer, V., Kipping J.,& Dijkstra, K. D. B. (n.d). Developing an Odonate-based tool for monitoring freshwater ecosystems in Rwanda. *Chapter 3 of doctoral dissertation. (A working paper in press).*

Verdone, M., & Seidl, A. (2016). How do the risk preferences of smallholder farmers affect the attractiveness of restoration activities? Evidence from forest and agricultural land in Rwanda. *Journal of Development and Agricultural Economics*, *8*(9), 200-214. DOI: 10.5897/JDAE2016-0739

Wang, G., Mang, S., Cai, H., Liu, S., Zhang, Z., Wang, L., & Innes, J. L. (2016). Integrated watershed management : evolution , development and emerging trends. *Journal of Forestry Research*, *27*(5), 967–994. https://doi.org/10.1007/s11676-016-0293-3

Watso, D.F., Philip, G.M. A., (1985). Refinement of Inverse Distance Weighted Interpolation . *Geoprocessing 2:*315-327.

Weigandt, M., Gyenge, J., Fernandez, M. E., Varela, S., & Schlichter, T. (2015). Afforestations and wetlands, are they a good combination? Study of water flues in two cases of Patagonian wetlands. *Ecohydrology*, *8*(3), 416-425. https://doi.org/10.1002/eco.1512

Wekundah, J.M. (2012). Why Protect Traditional Knowledge? *African Technology Policy Studies Network*, (44), pp.1–16. https://www.africaportal.org/documents/14996/sps44.pdf

Wilhere, G. F. (2002). Adaptive management in habitat conservation plans. *Conservation Biology*, *16*(1), 20-29. https://doi.org/10.1046/j.1523-1739.2002.00350.x

Wortley, L., Hero, J. M., & Howes, M. (2013). Evaluating ecological restoration success: A review of the literature. *Restoration Ecology*, *21*(5), 537–543.

https://doi.org/10.1111/rec.12028

Chapter 3: Developing an odonate-based tool for monitoring freshwater ecosystems in Rwanda

Abstract

Freshwater ecosystems are facing alarming threats of unsustainable resource use and development. In order to address these threats, there is a need to understand how these ecosystems are responding through the advancement of robust monitoring tools. This chapter presents an odonate-based tool for monitoring freshwater ecosystems in Rwanda, Dragonfly Biotic Index (DBI), developed and tested by sampling locations representing the major freshwater ecosystems of Rwanda, including streams and rivers, ponds and lakes, open savannah swamp and forest swamp, small seepages found in forests, and springs and similar freshwater habitats in both protected and unprotected areas. A total of 99 sites were visited in the short dry season, January through early March 2019, and revisited during the short rainy season, September through mid-November 2019. While habitat and environmental variables were directly measured in field. Adult odonates were sampled using a combination of observations at a distance and direct catch sampling with a sweep net. The DBI developed from these data consist of three sub-indices: Distribution-Based Score (DBS), Threat-Based Score (TBS) and Sensitivity-Based Score (SBS). The strength and convenience of DBI in ecosystem monitoring rests on the fact that it uses organisms that are not only sensitive to habitat change but also charismatic and relatively easy to identify. DBI is also useful in comparing both different locations and monitoring of a single habitat over time. A Habitat Integrity Index was determined based on data from sampled sites. The DBI had a strong correlation with the Habitat Integrity Index (HII) indicating the performance of DBI in reflecting habitat integrity. Additionally, this chapter identifies hotspot habitats for odonates in

Rwanda. These are defined based on species richness, presence of unique species and habitat integrity indicated by DBI site values. Habitats with high DBI site values in each ecological zone are suggested to be benchmarks for restoration. This study highlights the DBI as an accurate and precise tool to monitor freshwater ecosystems in space and time.

Key words: Odonate, dragonfly, biological indicator, ecosystem monitoring, freshwater ecosystem, habitat assessment, habitat integrity, restoration benchmark

Introduction

While freshwater ecosystems support a huge number of organisms and generate a wide variety of ecosystem services, they are facing alarming threats (Dudgeon et al., 2006 ; Turak et al., 2017). Freshwater ecosystems constitute less than 1% of the world's surface, and they harbor about 6% of all the world's known species (Dudgeon et al., 2006). There is accumulating evidence that the major threats to these ecosystems are human-induced (Dodds et al., 2013; Mangadze et al., 2019; Schmeller et al., 2018; Soesbergen et al., 2019). These threats are largely connected to land use conversion and pollution (Monteiro et al., 2015 ; Butchart et al., 2018). This, coupled with the predicted impacts of climate change, will particularly worsen conditions in freshwater ecosystems, if timely and effective interventions are not undertaken (Marques et al., 2018; Markovic et al., 2014; Taniwaki et al., 2017).

To tackle these threats, it is essential to track freshwater ecosystem responses to stresses. This requires robust ecological indicators that are not only optimized in accuracy and precision, but also sensitive to contemporary fast habitat changes. Based on this rationale, monitoring ecosystems using biological organisms, a concept known as bioindication, shows promise for efficient assessment of ecological integrity (Behn et al., 2018; Turak et al., 2017). Unlike traditional approaches that are based on physical and chemical parameters, which are constrained due to the limited range of responses captured at a single moment of sampling, bioindication operates on a broader spatio-temporal scale (Rocha-ortega et al., 2019). Bioindication has an elevated capacity for adequate analysis of the integrated responses of populations and communities of organisms as a whole. They also give insights for habitat states in the past, due to the fact that past events inherently shape the present biological indicators' community structures (Brito et al., 2018; Salmah et al., 2006; Mangadze et al., 2019).

In many African countries where freshwater ecosystems, such as wetlands, are being affected by both conversion to agricultural lands at a rapid rate (Cunha et al., 2019; Muñoz-Villers & López-Blanco, 2008), and climate change (Rebaudo & Dangles, 2015; Taniwaki et al., 2017), bioindication-based tools to monitor ecosystem functioning are urgently needed. These tools can generally increase cost-effectiveness and spatial specificity (Mangadze et al., 2019; Mendes et al., 2017; Parmar et al., 2016). While bioindication could be an answer where financial limitations are an issue, this practice is still lagging behind in most of African developing countries (Sayer et al., 2018). Furthermore, most of the efforts to apply bioindication for monitoring use techniques developed outside their ecological regions, making them less useful. Indeed, temporal and spatial variability in ecosystems amplified by both climate change and human development need to be accounted for in bioindication (Marques et al., 2018; Taniwaki et al., 2017). If we are to promote bioindication practices in developing countries, it is critical to develop tools that are practically appealing to the local communities intended to use them, as well as tailored to specific ecosystems of concern (Conrad & Hilchey, 2011; Ducarme et al., 2013).

The need to develop locally relevant ecological indicators is vital for adaptive management and restoration of ecosystems in developing countries of Africa; however, these indicators remain a challenge. While ecological indication techniques are fairly well understood among research ecologists and conservationists, they still need to be put into the hands of policymakers of African countries (Hartter & Ryan, 2010; Vaccaro et al., 2012). If policies were considering ecological data African countries would not be promoting practices that adversely affect the environment. These practices could be discouraged by negative feedbacks from ecosystems and motivate alternatives that are ecologically sustainable (Vaccaro et al., 2012). Not only are the lack of ecological indicators an issue, but adjustments of ecological indicators, when they exist, are not made prior to their application (Golfieri et al., 2016; Vorster et al., 2020).

The most adaptable commonly used ecological indicators include invertebrates. These have been instrumental in ecosystem assessment for decades (Siddig et al, 2016; Siziba et al., 2018). Invertebrates-based approach is more efficient when used at a lower taxonomic resolution (Berquier et al., 2016; Renner et al., 2016). For example, the use of odonata species, hereby referred to as odonates (insects that include two sub-orders: damselflies (zygoptera) and dragonflies (anisoptera) (Dolný et al., 2011; Samways, 2008)), demonstrates great appeal as practical and effective indicators of habitat integrity due to technical and logistical feasibility (Figure 1-3). Odonates are charismatic due to eye-catching colors, patterns and flying style (Maltchik et al., 2010; Mendes et al., 2017; Siddig et al., 2016; Simaika & Samways, 2018).

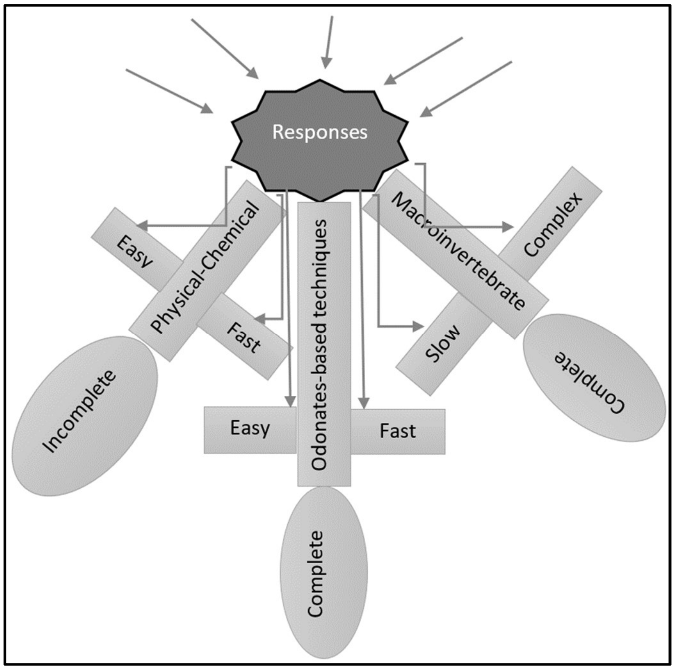

Figure 1-3. Conceptual model illustrating the reasoning for utilizing an Odonate-Based Index over physico-chemical or macroinvertebrate-based approaches. Odonates are relatively easier to learn and faster approach for bioindication than macroinvertebrate techniques. While physical-chemical-based approaches are fast and easy, they only capture a limited range of responses to stressors. Arrows above the "responses" symbolize stresses to ecosystems, while arrows below the "responses" show the detection of the responses.

The particularity of odonates as indicators lies in the following features: (1) Their rich number of species with varied tolerance to habitat disturbance, from generalist species with high tolerance to specialist species with low tolerance (McPeek, 2008; Valente-Neto et al., 2016). For example, Rwanda has a surface area of only 26,340 km^2 and has 114 known odonate species, which illustrates the size of odonate species richness (2) A large number of these species have a high "specificity" to habitats. This means the assemblages of these species tend to be abundant within habitats with a well-defined set of environmental conditions. Also, odonate occurrence has high fidelity to specific habitats, which means their occurrence is consistent to habitats with specific conditions (McGeoch et al., 2011).

I developed a Dragonfly Biotic Index (DBI) following the work pioneered in South Africa, as well as the Africa Dragonfly Biotic Index (Samways & Simaika, 2014; Simaika & Samways, 2009a; Vorster et al., 2020). The performance of DBI is evaluated based on its correlation with Habitat Integrity Index (HII). In this study, HII consists of environmental conditions of water bodies reflected in quality of riparian zone, types of land use and potential sources of pollutants (Luke et al., 2017; Monteiro-Júnior et al., 2014). I analyzed differences in odonate assemblages between seasons at several sites across ecological zones. Finally, I describe benchmark sites that can play a seminal role in restoration. Benchmarks are defined here based on DBI site values, species richness and presence of unique or endemic species.

Methods

Study Area

This study was conducted in Rwanda, a small (26,340 km^2) but highly diverse country in terms of ecosystems. One way to look at this ecosystem diversity is through differences in elevation, which is one of the major factors determining diversity in ecosystems. For example, the highest elevation is the Karisimbi volcano summit (4,507 meters) located in the northwestern part of the country, a region of afroalpine, alpine grasslands and afromontane forests. The elevation mostly has a gradual change. The lower elevations by Lake Kivu host gallery forest in the central and south west, 970 m elevation. From west to east, the high elevations of the mountains subside into rolling hills and marshy grassland valleys of the central plateau region. The gradual reduction in slope gradient extends toward the northeast and southeast with an area characterized by warmer savanna bushland. These blend into a landscape with broad river valleys, lakes, and papyrus swamp (Kindt et al., 2011).

Rwanda can be categorized into twelve agro-ecological zones based on characteristics such as elevation range and average yearly rainfall (Ford, 1990). Elevation (which influences temperature) and rain fall differently shape the soil and vegetation (Maltchik et al., 2010) and are major limiting factors for odonate assemblages. I reclassified the twelve agro-ecological zones into six categories referred to as ecological zones (Table 1-3), based on elevation, in order to better capture the major patterns and attributes important in odonate species distribution at country scale.

Table 1-3: The major variables that define the six ecological zones. This highlights the range of elevation in each ecological zone, average of yearly rainfall, soil and number of sample sites in each ecological zone Ford (1990).

Ecological Zones	Elevational ranges (m)	Average rainfall/year (mm)	Soil Types	Number of Sample sites
South West	970-2500	1200-1500	Oxisols, alluvial and heavy basaltz	25
North West	1400-4500	1200-1600	Volcanic soils and superficial loamy clay	9
South Central	1350-1700	1050-1200	Ultisols, clay, schist and humic soil	17
North Central	1900-2300	1100-1200	Ultisols, high altitude lateritic	18
South East	1400-1800	900-950	Ultisols, oxisols, and altered clay	12

North East	1250-1600	850-900	Ultisols, oxisols, and old variable soil	18

Odonate sampling

This study examines the distribution and habitats of odonate species throughout the ecological zones of Rwanda (Table 1-3), with the goal to develop a Dragonfly Biotic Index (DBI). The index consists of three sub-indices: Distribution-Based Score (DBS), Threat-Based Score (TBS) and Sensitivity-Based Score (SBS)of recorded odonate species (Table 2-3). The developed DBI in this study was modeled from similar work in South Africa (Samways & Simaika, 2014; Simaika & Samways, 2009a). While the three DBI sub-indices of this present study in Rwanda and earlier one in South Africa are fairly identical, there are slight differences in each given the differences in the scale and ecosystems. The main difference is in the third sub-index, TBS, whereby the scores assigned to categories of IUCN Red List in Rwanda don't all match with those of South Africa.

To calculate DBI, a total of 99 sites were visited. Sample sites were purposely selected to be representative of the major freshwater ecosystems of Rwanda, consisting of streams and rivers, ponds and lakes, open savannah swamp and forest swamp, small seepages in forest, springs and similar freshwater habitats in both protected and unprotected areas. All sites were in close proximity to trails or roads, as accessibility was a factor in site selection.

The sampling was seasonal. Among the four seasons of Rwanda: long and short rainy season, and long and short dry season (Ntwali et al., 2016; Mukanyandwi et al., 2019), odonates were sampled during the short dry season, January through early March 2019, and revisited during the short rainy season, September through mid-November 2019. The average of precipitation and air temperature during the two shorter seasons are representative of the two other longer seasons (Ntwali et al., 2016). To collect species, at each of the 99 sites sampled, odonate adults were collected for one hour when just myself was sampling, or 0.5 hours when myself and a field assistant were sampling. Sampling was conducted between 09 am and 5 pm, only when the weather was sunny and wind was at a minimum (wind speed ≤8km/h) with temperatures above 19° C; odonates decrease their activity below this temperature (Dutra & Marco, 2015). Adults of odonates were collected or observed along a reach of 100 m. By walking back and forth along one bank of the water channel, any species observed within 10 m perpendicular to water body was caught using a sweep net if possible, identified in the field following the field handbook of Dijkstra and Clausnitzer (2014) and kept in paper envelopes. Thanks to the department of Tourism and Conservation of the Rwanda Development Board for the permit for these collections. When collection was not possible, I used a combination of observations at a distance with naked eyes or binoculars at distance when details cannot be observed by naked eyes. Species recorded were either flying or perching in the middle or above water body or riparian zone. Collected specimens were washed in acetone after every day of field work. A sample point for other variable measurements (for water physical-chemical, environmental and habitat characteristics) as well as GPS coordinates (using WGS_1984 datum) was placed in the middle of the 100 m stretch (Walsh et al., 2007).

Development of Dragonfly Biotic Index

Dragonfly Biotic Index (DBI) is a tool to assess ecological conditions or habitat integrity based on odonate assemblages. It is meant to assign a score to each of the species that inhabit a site, then scores of all species collectively reflect the conditions of a site (habitat integrity). The score that is assigned to each species consists of three sub-scores: Distribution-Based Score (DBS), Threat-Based Score (TBS) and Sensitivity-Based Score (SBS). The three scores constitute DBI for each species (Appendix 1-3). DBI score for an individual species is the sum of each species' Distribution-Based Scores (DBS), Threat Based-Scores (TBS) and Sensitivity-Based Scores (SBS) and it ranges from 0 to 9.

Distribution-based score (DBS): Each of the recorded species was given a sub-score ranging between 0 and 3 according to its distribution across ecological zones of Rwanda. Species that are common and widespread throughout the six ecological zones receive the lowest score (0). Species that are common but not found in all ecological zones are ranked 1. Species that are given a sub-score of 2 are those found in three ecological zones at most, while those that are endemic to the country and found only in one or two ecological zones are given the highest sub-score (3).

Sensitivity-based score (SBS): This sub- score, which ranges between 0 and 3, is based on criteria typically identified as characteristic of good indicator species, fidelity and specificity. Species that are scored 0 are those that are tolerant to disturbed, polluted, degraded and/or artificial habitats and/or can be found where alien plants are present (Samways & Simaika, 2014; Simaika & Samways, 2009a). If either or all of these habitats constitute more than a third of the habitats where a species was found during sampling, that species is considered to be the least sensitive and not fully meeting the fidelity condition. Species with a score of 1 are considered of low sensitivity to habitat disturbances. These are species for which one third or less of the habitat they were found in is artificial, degraded, disturbed, polluted, and/or with alien plants present. Species of medium sensitivity (a score of 2) are not found in any artificial water bodies, but other habitats similar to the species of low sensitivity. The highest score (3) are for those species that are extremely sensitive and only recorded in intact natural habitat. These species exhibit a high specificity to such habitats(Samways & Simaika, 2014; Simaika & Samways, 2009a)..

Threat-based score (TBS): TBS is score built on categories of the IUCN red list (Samways & Simaika, 2014; Simaika & Samways, 2009a). In addition to species commonness as reflected by DBS and sensitivity score from the SBS, the TBS adds another value layer based on extent to which more attention for conservation is needed. The following scores: 0, 1, 2 and 3 are associated with the following IUCN red list categories, respectively: Least Concern, Near Threatened, Data Deficient/Vulnerable, Endangered/Critically endangered (Samways & Simaika, 2014; Simaika & Samways, 2009b). Given the lack of information on the real status of species categorized as Data Deficiency on IUCN red list, these species are grouped with the middle category in the IUCN red list spectrum and scored the same as "Vulnerable". The species in the Data Deficiency category have the potential to be up-listed to Endangered category or down listed to Near Threatened as data become available. The Endangered species are grouped together with Critically Endangered species to acknowledge the risk to be critically endangered due to pressure they are particularly confronted with in Rwanda as related to human density. New species that are not yet listed on IUCN are scored as critically endangered until studies on their population prove otherwise. The score for a new species is set conservatively rather than let it be at risk of extinction if the habitat is not protected.

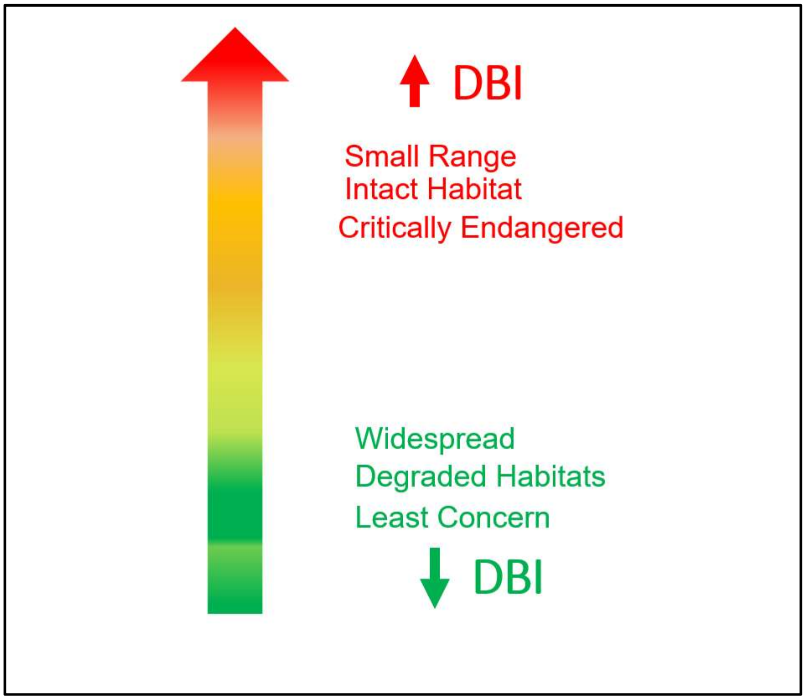

Figure 2-3: DBI scoring spectrum: Species that are assigned the highest scores are those that are restricted to small geographical (high DBS), only found in intact habitats i.e very sensitive (high SBS), or critically endangered (high TBS). Species on the other end of the spectrum is assigned highest score.

Table 2-3: Calculating the Dragonfly Biotic Index (DBI)

Scores **Sub-indices**	**0**	**1**	**2**	**3**
Distribution-Based Scores (DBS)	Species that are very common & widespread	Species that are widespread but not found in all ecological zones	Species that are found in not more than three ecological zones	New species or endemic in one or two ecological zones

IUCN, Threat-Based Scores (TBS)	Least concern species	Near threatened species	Data deficient or Vulnerable species	Endangered or critically endangered or new species
Sensitivity-Based Scores (SBS)	Species with more than a third of their habitats composed of either alien plants, or disturbed habitat (with signs of human activities) banks, or artificial water bodies	Species that are not found in all disturbed banks present at the site; Scarce (equal or less than a third of records) in artificial or disturbed water bodies	Species that are not found in artificial (created pools, dams or ditches) water bodies	Species that are found only in undisturbed habitats

Overall DBI Site Scoring

DBI score for an individual species is the sum of each species' Distribution-Based Scores (DBS), Threat Based-Scores (TBS) and Sensitivity-Based Scores (SBS) and it ranges from 0 to 9. The DBI value for each site is the sum of all DBI scores of all species divided by the total number of species (N) recorded within a site (Equation 3). "DBI_1+DBI_2+DBI_3+… DBI_N", where species are represented by 1, 2, 3, to N.

$$\textbf{DBI site value} = \sum_{n=1}^{N} \frac{DBI_1 + DBI_2 + DBI_3 + \cdots + DBI_N}{N} \quad \ldots\ldots\ldots\ldots \text{(Equation 3)}$$

The calculation of DBI (equation 3) provides a value at each surveyed habitat. While the DBI score (DBI_1 + DBI_2_+DBI_3…DBI ($_N$)) increases with species richness of a habitat (N), the maximum of DBI site value is 9, since all the species DBI scores are divided by the number of species (N), as explained by the equation 3.

Habitat Integrity Index

The habitat integrity index (HII) consists of scores assigned to a set of variables that are considered factors reflecting habitat condition (Luke et al., 2017; Monteiro-Júnior et al., 2014) These factors take into account human activities, such as cropland, dump sites, mining sites, buildings and domestic or industrial wastes.

Riparian vegetation included *Arundinaria alpine* and *Pennisetum purpureum* usually planted in Rwanda to support riparian zones among other purposes. Natural vegetation in riparian zones consisted of *Cyperus papyrus, Echinochloa pyramidalis*, *Phragmites mauritianus, Phoenix reclinata* and *Typha latifolia.* Human occupation refers to presence of buildings and other infrastructure. Site with less than one building per 60 m^2 was considered “non-dense” and “dense” otherwise. As for mining and dump, site scoring was done based on the proximity of a dump or mining site to a water body and whether they are active or inactive. Mining sites consisted of open land mining (e.g. sands, gravels, and rock extractions and other minerals). Trash in the dump sites ranges from plastic, metal, glass, rubber, building materials to organic matters. Domestic and industrial wastes were scored based on the presence of the number of effluents the water body. Croplands are defined based on crop density or spacing; crops <2 meters from each other were considered dense.

The index is based on principles that habitats of high integrity are those with minimum impacts from humans (Miguel et al., 2017). These habitat factors are further broken down into degree and proximity of the human activity to the water bodies (Table 3-3).

Table 3-3: Scores for Habitat Integrity Index.

Scores Variables	**0**	**1**	**2**	**3**
Riparian (RV) Vegetation	Absence of riparian vegetation within 10 m of the bank of water body	Presence of plants meant to support riparian zone within 10 m	Natural riparian vegetation but not intact within 10 m	Natural protected intact riparian vegetation within 10 m (within a protected area)
Cropland (CR)	Dense crops within 10 m distance from the sample point	Spaced crops within 10 m distance from the sample point	Presence of plants meant to hold soil within 10 m or spaced crops beyond 10m distance from water body	Absence of crops and presence of land in fallow within 10 m distance from the sample point

Dump Site (DS)	Presence of dump sites within 10 m	Presence of inactive dump sites within 10 m	Presence of inactive dump sites or restored beyond 10 m	Absence of dump sites in the upstream area
Mining Site (MS)	Presence of mining site within 10 m	Presence of inactive mining site within 10 m	Presence of inactive sites or under restoration process site beyond 10 m	Absence of mining sites in the upstream area
Human Occupation (HO)	Presence of dense buildings within 10 m	Low density of buildings within 10 m	Presence of buildings beyond10 m	Absence of buildings in the upstream area within 500 m
Domestic or Industrial Wastes (DIW)	Presence of at least 3 domestic or industrial effluents within 500 m of upstream	Presence of 2 domestic or industrial effluents within 500 m of upstream	Presence of 1 domestic or industrial effluents within 500 m of upstream	Absence of domestic or industrial effluents within 500 m of upstream

The scoring of HII consisted of an array of habitat characteristics ranging from the well-preserved or intact habitats (score = 3) to disturbed habitats (score = 0) (Table 2-3). Scores for each of these six characteristics were added for each site, and divided by 18 to create a site HII for each site ranging from 0 to 1.

This was modeled after the method used by Monteiro-Júnior et al. (2014), which is described as follows:

$$\boldsymbol{S_i = \frac{V_i}{18}} \qquad \text{(Equation 1)}$$

where "S_i" is the weighted score for the ith variable of habitat integrity, "V_i" is the score recorded for the variable, and "18" is the maximum possible score for the variable. The S_i values are then used to calculate the HII, which include 6 variables, the HII for each site is the sum of all scores divided by the maximum possible score “18”:

(RV)/18+(CR)/18+DS)/18+MS)/18+HO)/18+DIW)/18. These are summarized as follows:

$$\boldsymbol{HII} = \sum_i^6 \boldsymbol{S_i} \qquad \text{(Equation 2)}$$

where “6” is the number of included variables and i represents each single variable included.

Analysis

To understand the difference in odonate species abundance between wet and dry seasons across ecological zones, I used contingency tables and Chi-square tests in R (R Core Team 2020) with the MASS package (Ripley, 2019). I used the Rmisc package (Hope, 2013) to plot the overall average of DBS, SBS and TBS. I conducted Spearman's rank correlation in R with ppcor package (Kim & Kim, 2015) to analyze the relationship between the DBI and HII. I used the ggplot package (Wickham, 2016) for the correlation and averages, Figures 4-2 & 6-2. I classified sites based on HII scores. Sites with HII < 0.4 are highly impacted (very disturbed), HII ≤ 0.4 < 0.8 are medium impacted (medium disturbance), while those with HII ≥0.8 have not been impacted (no disturbance).

Based on DBI and HII values, I identified habitats that could be considered as hotspots for odonates and benchmarks for restoration. An odonate hotspot was considered a habitat that had at least one unique species, not yet recorded from any other site in the country (Appendix 2-3) and/or had more than 20 species recoded (Figure 3-3). A value of 20 was selected because it is double the average of species richness recorded in all sites during this study. To identify benchmarks for restoration, I identified habitats with DBI site value ≥3.5. This DBI site value is considered to be high enough to represent habitat with good ecological conditions since it coincides with an HII value of 0.875. This value (HII=0.875) falls within the range of scores that reflects habitats that do not have human impacts and are relatively intact (Figure 3-3). These habitats could therefore play a benchmark role for restoration (Table 5-3).

Results

Species checklist

The countrywide survey recorded 91 odonate species. This survey along with prior surveys (Clausnitzer et al., 2011; Kipping et al., 2017; Paulson, 2011) brought the total number of odonate species recorded in Rwanda to 114 (Appendix 3-3) . This includes 25 new species to the national checklist, added by this study. The average species richness and abundance in all sites was 10 (range = 1 to 28) and 46 (range = 2 to 181), respectively.

A comparison of abundance of odonate species between seasons and ecological zones The analysis of species in rainy and dry seasons across ecological zones shows a significant difference in species abundance between ecological zones and between seasons ($X^2 = 110.04$, df = 5, p-value<0.001; Figure 3-3).

Summary of sub-indices constituting the DBI

I present a list of the sub-indices (DBS, SBS, TBS) for 91 species sampled in this study (Appendix 1-3). To reveal status of odonate species in terms of their sub-indices, I calculated the average of each sub-index (DBS, SBS and TBS) recorded in all sites in order to get a sense of the influence of each of them on DBI. The average of DBS appears to be the highest, followed by SBS and TBS (Figure 5-3).

Additionally, I analyzed the frequency of scores for each sub-index, DBS, SBS and TBS. I found that 50.87% of all species sampled had a DBS=1 and 50.56% have SBS=1, while 10.01% had a TBS=2 and 97.65% of species sampled had a TBS=0 (Figure 6-3). Almost half of the species sampled are widespread but not found in all ecological zones. As for SBS, about a half of species sampled are tolerant but not found in all disturbed habitats, while TBS results show that most of species are of least concern in terms of IUCN Red List.

For conservation and restoration purposes, I present a map of species richness across the country and list hotspot habitats for odonates (with high richness, high DBI site value and/or presence of unique species ((Figure 3-3) & (Appendix 2-3)). I identified sites with the highest DBI site value in each ecological zone and sites with DBI site value ≥3.5, with the objective to identify sites that can play a reference role for restoration, which are referred to as benchmarks for restoration (Table 5-3).

Regarding the relationship between HII and DBI, I found a strong positive correlation between Dragonfly Biotic Index and Habitat Integrity Index (Spearman's rank correlation, p-value <0.001, r=0.448; Figure 7-3).

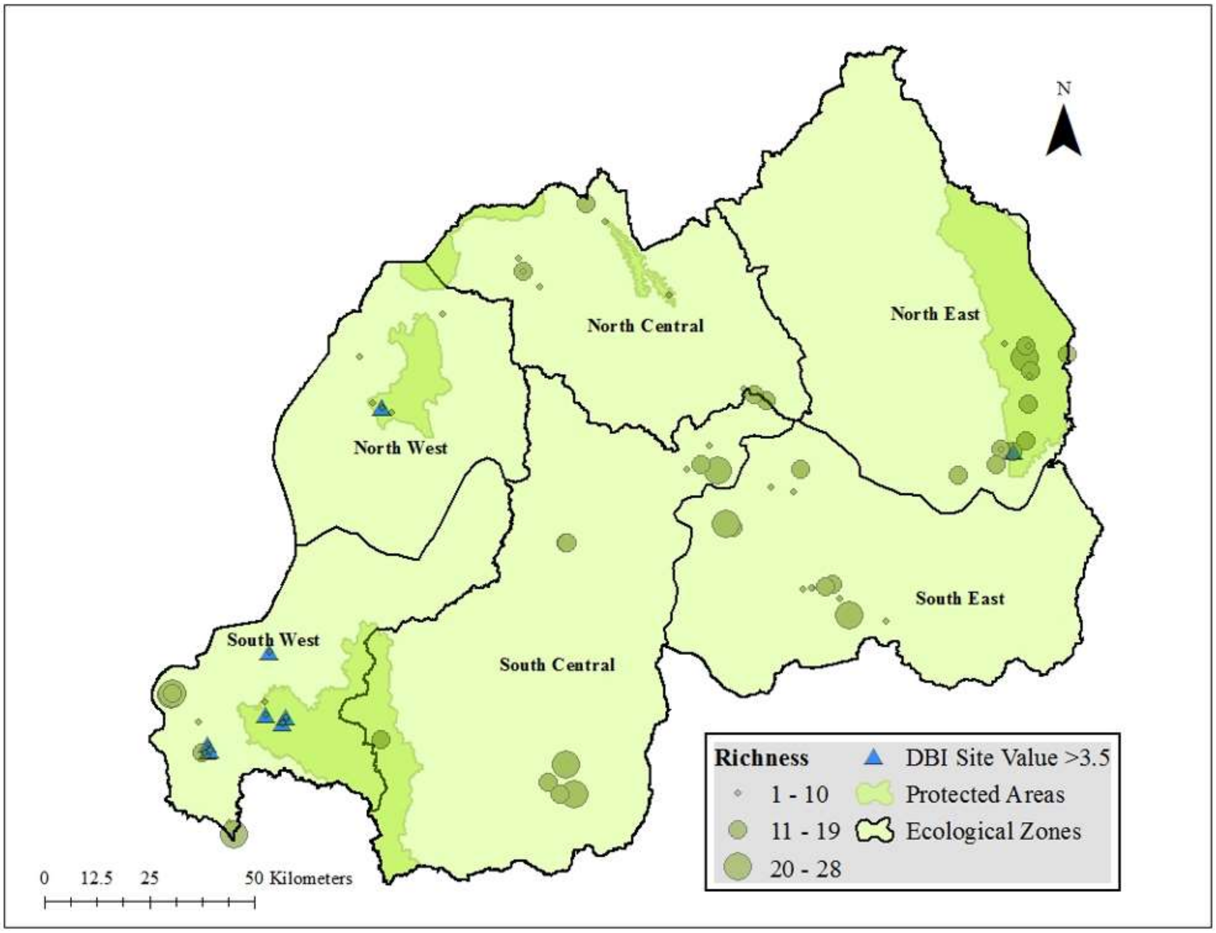

Figure 3-3. Study sites and odonate species richness per ecological zone in Rwanda, as well as sites with high DBI site value.

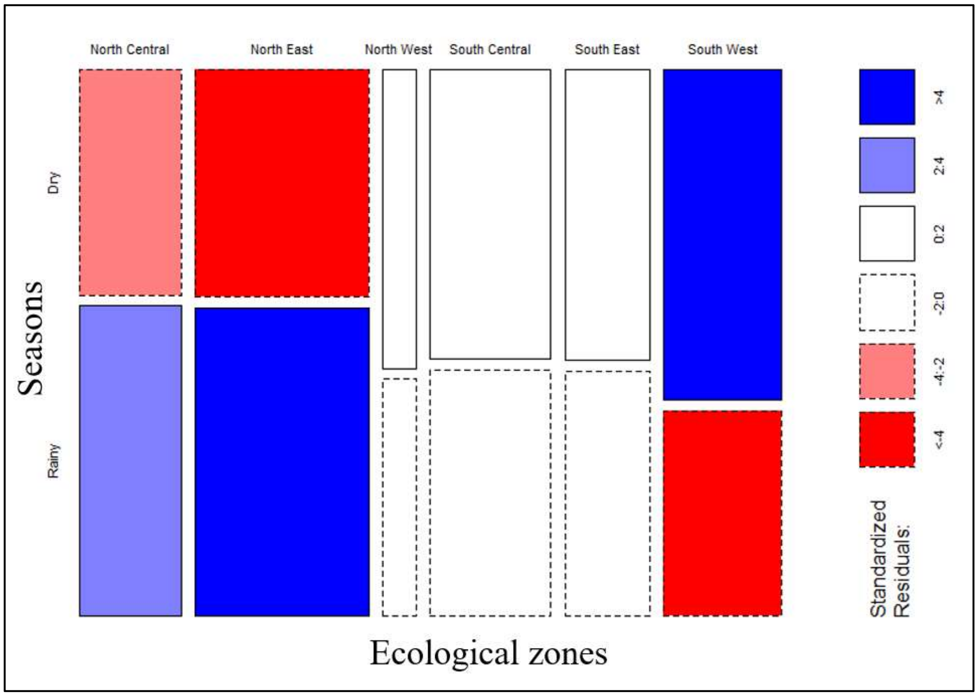

Figure 4-3. A comparison of abundance of odonate species between seasons and ecological zones. Darker colors (red or blue) indicate a greater difference in abundance between seasons and/or ecological. Solid lines mean the difference in abundance between seasons and/or ecological zones is greater than expected, while dotted lines indicate that the differences in abundance between ecological zones and/or seasons is less than expected.

Table 4-3: Standardized residuals of differences of odonate abundance between ecological zones and seasons. Positive values (in bold) show where the abundance is higher than expected, while mean values are abundances that are less than expected.

Ecological Zones	**Dry**	**Rainy**
North Central	-3.246729	**3.285597**
North East	-4.073673	**4.122441**
North West	**1.165615**	-1.179569
South Central	**1.507516**	-1.525563
South East	**1.352744**	-1.368938
South West	**4.665738**	-4.721593

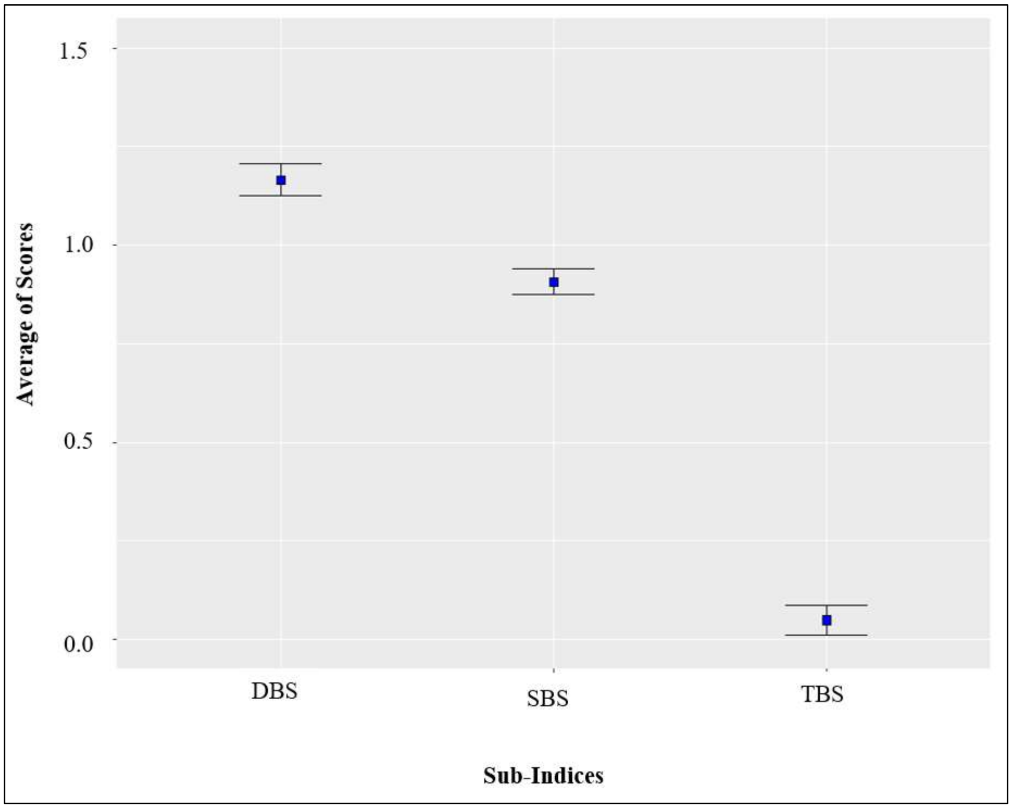

Figure 5-3. Average of scores for each of the sub-indices: Distribution-Based Score (DBS), Sensitivity-Based Score (SBS) and Threat-Based Score (TBS), in all recoded species. The x axis represents the sub-indices, while y axis is the average score for each of the sub-indices.

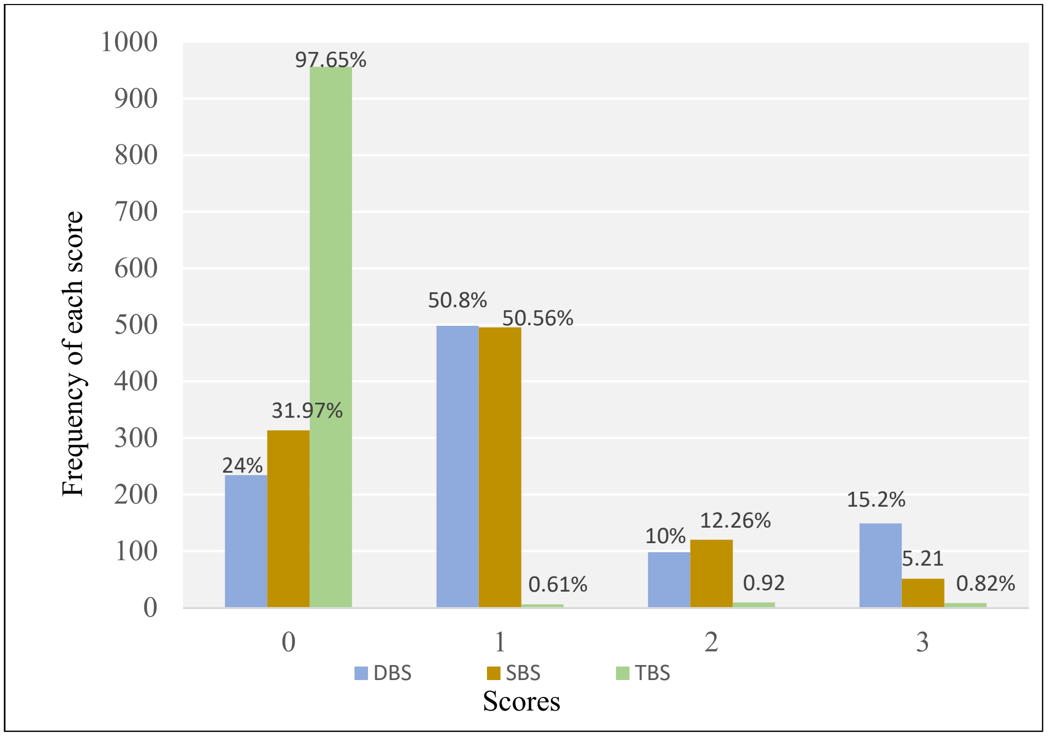

Figure 6-3. Frequency and percentage of scores for each of sub-index, Distribution-Based Score (DBS), Sensitivity-Based Score (SBS) and Threat-Based Score (TBS). For example, species whose DBS =3, represent 15.2% of all recorded species. Species whose SBS=3 are 5.21% of all recorded species, while those with TBS =3 represent 0.82% of all recorded species

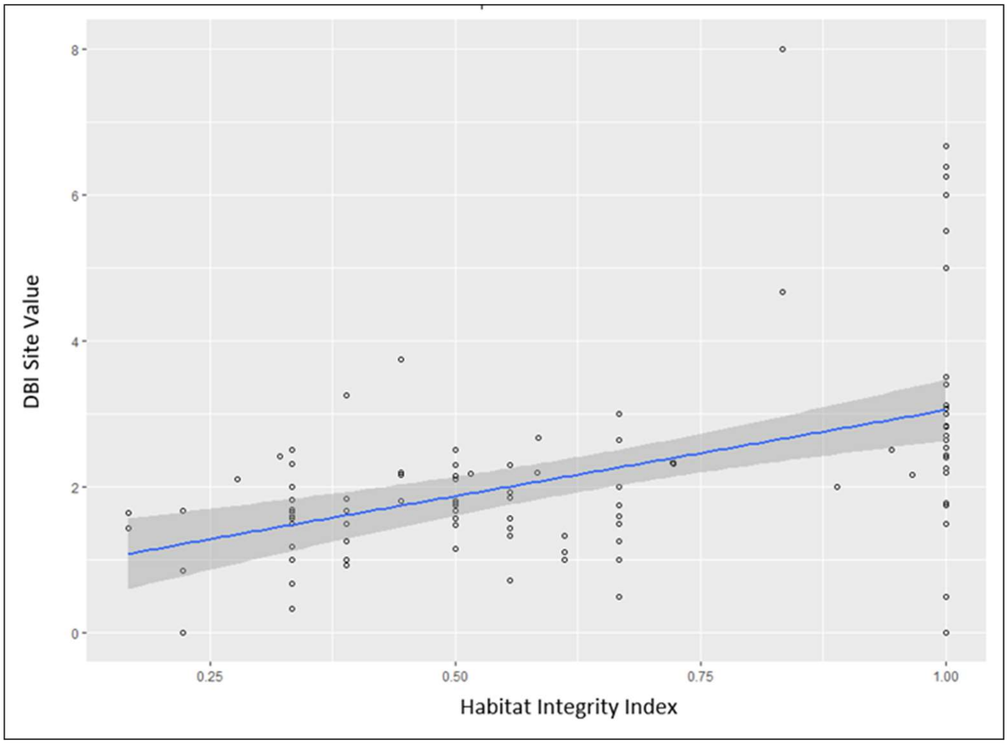

Figure 7-3. Relationship between Habitat Integrity Index (HII) and Dragonfly Biotic Index site value (DBI site value)

Table 5-3: Identifying restoration benchmarks based on sites with high DBI Site in each zone. The sites with (*) are habitats that can play a role of restoration benchmark (DBI site value≥3.5). Note: The site of highest DBI in North Central has a DBI ≤3.5, thus not high enough to play a reference role.

Ecological Zone	Site	DBI site Value
South West	Nyungwe-Karamba stream*	6.38
North West	Pfunda stream*	6.67
North East	Akagera, papyrus swamp*	4.67
South Central	Mwange stream*	3.5
South East	Akagera river wetland	2.5
North Central	Masaka pond	2

Discussion

The Dragonfly Biotic Index (DBI) was developed based on species recorded across six ecological zones of Rwanda. Each species was assigned a Distribution-Based Score (DBS), Sensitivity-Based Score (SBS) and Threat-Based Score (TBS), which together constitute the DBI for each species. DBI site values guide assessment of ecological conditions of habitats, as represented by a collection of DBI of each species within a habitat. Here, I discuss the strength of DBI validated through its congruence with the Habitat Integrity Index (HII), the use of DBI across ecological zones as a practical tool as well as an easy step-by-step guide on how to use the DBI.

Congruence of Dragonfly Biotic Index and Habitat Integrity Index

Well selected indicators have the potential to reveal clearer and simpler information from complex ecosystems (Fu et al., 2019; Parmar et al., 2016). The type of indicator used depends on the objectives of an assessment, and may include assessments of climate trends, environmental changes, community diversity, environmental chemistry, and habitat quality; these are referred to here as parameters of habitat integrity (Parmar et al., 2016). The Dragonfly Biotic Index developed here seeks to capture ecological responses to habitat disturbance that affect the habitat integrity. The efficiency of this index is emphasized by the strong positive correlation found between DBI and the Habitat Integrity Index (HII). The primary variables that govern the HII include both the presence and state of riparian vegetation, and proximity to sources of pollutants. In this study, the sources of pollutants included mining, agriculture and domestic and industrial wastes, which are also the main threats to riparian vegetations, an important part of freshwater systems (Monteiro-Júnior et al., 2014; Miguel et al, 2017). The importance of riparian vegetation to filter pollutants, support food webs and regulate temperature has been well documented (Behn et al., 2018). The relationship of DBI to HII underscores the performance of DBI given the strong association between HII and healthy ecosystem functioning (Behn et al., 2018). While functional and services of habitats are considered key parameters for habitat integrity (Rabeni, 2000), physical structure and habitat intactness from human impacts are also important and quantifiable attributes for habitat integrity (Caniani et al., 2016; Gerson et al., 2003).

The contributions of the sub-indices (DBS, SBS and TBS) on the DBI and implications for conservation

The DBI approach is consistent with earlier work that outlines the importance of avoiding the use of just one indicator species (Alsterberg et al., 2017; Villéger, 2008). One indicator species operates under linear or one-dimensional assumptions, which can skew results for habitat bioindication. However, designing an index that maximize capturing responses to degradation is a daunting challenge (Villéger, 2008). To turn around these shortcomings, the use of an entire community of species within a habitat may optimize the accuracy of bioindication in characterizing ecological integrity within a habitat (Berquier et al., 2016; Miguel et al., 2017). The DBI seeks as many facets as possible to characterize a habitat by using odonate community sub-scores that reflect the status of threats to habitat. This index integrates three sub-indices based on information about each species present in the community. The strength of the DBI rests on the fact that it provides an evaluation of the state of habitat integrity and gives a sense of conservation value. These are weighed through the three axes that comprise the DBI: Distribution-Based Score, Sensitivity-Based Score and Threat-Based Score.

The three sub-indices jointly contribute to revealing sites of conservation priority. The averages of the sub-indices show that the Threat-Based Scores (TBS) for all recorded species in Rwanda is considerably lower than sensitivity-Based Scores (SBS) and Distribution-Based Scores (DBS). The TBS could be regarded as less influential to the DBI due to its overall lower score and thus less powerful in its ability to reveal habitat threats that call for special conservation attention. It is worth pointing out that national Red List could increase the TBS scores. For example, species restricted to narrow ranges (Appendix 2-3) could be categorized as nationally threatened or endangered, while many of them are widespread outside Rwanda. On one hand, using national Red List information could cause the TBS scores to be higher. If the TBS is based on distribution ranges, more species could be listed as critically endangered, threatened or vulnerable at national level. Therefore, this could increase the number of sites that need special conservation attention at national level. On the other hand, continental or global IUCN Red List gives more room for further development of standard indices at a wider scale beyond national boundaries (such as regional or continental), which is the reason this study chose to use the global IUCN Red List.

The results from the analysis of sub-indices frequency, unsurprisingly, indicate that almost half of the species sampled are widespread but not found in all ecological zones. The same results highlight that half of the species observed are tolerant to disturbance, but not found in all disturbed habitats. The results show that most of the odonates sampled in this study are of Least Concern in terms of IUCN Red List. It is worth noting that some of the sub-indices are correlated. Sites with higher TBS are likely to have a higher SBS and DBS, indicating a site with high conservation priority.

The correlation between sub-indices is seen, for instance, in the highest DBI site value in the country, Karamba stream in Nyungwe National Park, which harbors species whose geographic distribution is restricted to one ecological zone, making the Distribution-Based Score to be higher. These species are only found in habitats with no or a minimum disturbance, i.e. high Sensitivity-Based Score, or species with high IUCN Red List value (high TBS). For example, one species that contributes to the Karamba DBI site value is *Pseudagrion kamiranzovu*, which has a restricted geographic distribution as it is only found in one ecological zone, only in undisturbed habitats, and Red Listed as least concern by IUCN. *Stenocypha jacksoni* is yet another species that contributes to the Karamba DBI site value, as its DBS (3/3), SBS (3/3) and TBS (2/3) are all high. The sub-indices are not always correlated. Exceptions include, for example, *Atoconeura eudeudoxia*, found at the same site as the above species and also geographically restricted within only two ecological zones and inhabits only intact habitats; however, it is listed as least concern on the IUCN Red list since its population is widespread outside the region. This is the same case for *Afroaeschna scotias*, a species scored high for its national distribution and sensitivity but least concern for the global IUCN Red List.

Accounting for seasonality and location specificity in DBI-based monitoring

Rwanda ecological zones vary along longitudinal and latitudinal gradients. Along ecological zone gradients there are contrasts in species richness. In part, changes in species richness across ecological zones is due to differences in precipitation and temperature (Table 5-3). Precipitation and temperature are the major factors determining seasons, and odonates can display seasonal variation in their populations (Majer et al., 2013). The explanation for this is that the rainy season sustains more odonate habitats. The rains fill depressions, creating vernal pools, and expand rivers and lakes when the water table overfloods. This is consistent with results showing differences in abundance between seasons and ecological zones (Table 4-3

Most contrasts between ecological zones are associated with a difference in average temperature, annual precipitation, and soil between specific sites and ecological zones as whole. For example, the South East ecological zone has higher relative abundance than expected in the dry season, compared to the rainy season, which may be due to the consistent low average annual precipitation and moderate temperature across seasons in this zone (Ndayisaba et al., 2016). The soil is high in clay in this region, which supports water retention during and after rain, as opposed to the North East zone where soils have less clay (Habarurema & Steiner, 1997). The soil type supports the lack of variation in odonate abundance between seasons in the North West zone, where the soil is predominantly volcanic which is porous (Lu et al., 2018; Romero et al., 1999). This creates less difference in water body quantity and distribution between seasons.

In addition to soil types both average seasonal temperature and precipitation are important factors that determine plant community composition and distribution and in turn shape water chemistry, all of which impact species colonization (Pereira et al., 2019). Therefore, seasonality and geographic location (site specificity) need to be accounted for when establishing monitoring programs. In order to optimize the accuracy of overall DBI site values, DBI-based monitoring should cover at least two seasons and should be site specific, given spatio-temporal variability (Samways & Grant, 2007).

How to Use the Odonate Based Tool (DBI) in Monitoring Ecosystems

The DBI should be applied in habitat monitoring based on insights from the present study as well as previously published studies from other locations. The DBI is a reliable tool for assessing freshwater habitat integrity and monitoring restoration progress (Samways & Simaika, 2014). It is based on observations of adults, both males and females. It can used for both running and stagnant water bodies. The DBI site value provides a way to compare localities. DBI could be used to compare sites of interest with relatively pristine sites or reference sites. This could inform to what degree the sampled sites differ from each other in terms of ecological integrity. DBI also provides a means to evaluate a site over time, when the program goal is long-term monitoring.

Here, I present an example of how to compare sites through DBI site values. The calculation for DBI site values, comparing Nyamabuye stream of Cyamudongo forest and Pfunda stream of Gishwati Forest, follows equation #3 above.

Table 6-3: Example of calculation of Dragonfly Biotic Index for two sites in Rwanda

Sites	**Species DBI: DBS+SBS+TBS**	**Total DBI Scores**	**DBI site values=Total DBI/Richness**
Nyamabuye stream of Cyamudongo forest	*Notogomphus lujai*:3+3+0=6 *Pseudagrion spernatum*:1+0+0=1 *Orthetrum camerunense*:1+1+0=2	6+1+2=9	9/3=3
Pfunda stream of Gishwati Forest	*Stenocypha tenuis*:3+2+0=5 *Atoconeura pseudeudoxia*:3+3+0=6	5+6=11	11/2=5.5

While this example compares just two sites from two different localities, in principle, there should be at least five site replicates within a locality to make comparisons statistically sound. It is recommended that the compared localities have the same number of replicates. The comparison of different time periods for the same locality requires multiple samplings. For example, if a degraded locality is under restoration, at least three samplings should be undertaken for a determined duration (for example, three consecutive days). Then, three more samplings should occur for the next time period of the same duration, and so on. This sampling pattern could be repeated over several years to monitor change. It is highly recommended to consistently stick to one season while monitoring or assessing localities. Otherwise, covering all seasons, when time and means permits is recommended.

Conclusion

- The DBI developed for Rwanda provides ecologists, environmental decision makers and local communities with a robust monitoring tool for assessing freshwater habitats and a method to prioritize sites for special conservation and restoration.
- DBI is a potentially useful tool for citizen science and environmental education programs as it is easy for the layperson or youth to learn.
- I propose the inclusion of DBI in all habitat monitoring and assessment programs. These include environmental impact assessment programs, restoration programs as well as prioritizing sites that need special attention.
- It is recommended to account for differences in seasonality and ecological zones when designing the monitoring plan. This means that the comparison of localities should take place within the same season, especially when it is not feasible to sample in all seasons. Comparisons are more effective if the localities in question are within the same

ecological zone. The consideration of seasons and ecological zones applies while monitoring single localities as well. For this, I provide a list restoration benchmarks in Rwanda as reference against which to compare localities within each ecological zone.

- To increase the accuracy and applicability of this tool, more field surveys are needed to uncover species that have not yet been recorded, and data are needed from long dry season (June-August) which was not covered in this study.
- Finally, given the strong interconnection and transboundary nature of freshwater systems in Africa, I recommend the development of similar indices tailored to other African regions (using local species of odonates), in order to make DBI a standard monitoring technique synchronized across the continent.

Reference

Alsterberg, C., Roger, F., Sundbäck, K., Juhanson, J., Hulth, S., Hallin, S., & Gamfeldt, L. (2017). Habitat diversity and ecosystem multifunctionality—The importance of direct and indirect effects. *Science Advances*, *3*(2), 1–10. https://doi.org/10.1126/sciadv.1601475

Behn, K., Becker, M., Burghof, S., Möseler, B. M., Kyalo, D., & Alvarez, M. (2018). Using vegetation attributes to rapidly assess degradation of East African wetlands. *Ecological Indicators*, *89*(January 2017), 250–259. https://doi.org/10.1016/j.ecolind.2018.02.017

Berquier, C., Orsini, A., Ferrat, L., & Andrei-ruiz, M. (2016). " Odonata Community Index – Corsica " (OCIC): A new biological index based on adult odonate populations for assessment of the ecological status of watercourses in Corsica. *Ecological Indicators*, *66*, 163–172. https://doi.org/10.1016/j.ecolind.2016.01.022

Brito, J. G., Martins, R. T., Oliveira, V. C., Hamada, N., Nessimian, J. L., Hughes, R. M., … de Paula, F. R. (2018). Biological indicators of diversity in tropical streams: Congruence in the similarity of invertebrate assemblages. *Ecological Indicators*, *85*(October 2017), 85–92. https://doi.org/10.1016/j.ecolind.2017.09.001

Butchart, S. H. M., Walpole, M., Collen, B., Strien, A. Van, Jörn, P. W., Almond, R. E. A., … Lamarque, J. (2018). Global biodiversity: indicators of recent declines. *Science*, *328*(5982), 1164-1168. https://doi.org/10.1126/science.1187512

Caniani, D., Labella, A., Lioi, D. S., Mancini, I. M., & Masi, S. (2016). Habitat ecological integrity and environmental impact assessment of anthropic activities: A GIS-based fuzzy logic model for sites of high biodiversity conservation interest. *Ecological Indicators*, *67*, 238–249. https://doi.org/10.1016/j.ecolind.2016.02.038

Clausnitzer, V., Dijkstra, K.-D. B., & Kipping, J. (2011). Globally Threatened Dragonflies (Odonata) in Eastern Africa and Implications for Conservation. *Journal of East African Natural History*, *100*(1–2), 89–111. https://doi.org/10.2982/028.100.0106

Conrad, C. C., & Hilchey, K. G. (2011). A review of citizen science and community-based environmental monitoring: Issues and opportunities. *Environmental Monitoring and Assessment*, *176*(1–4), 273–291. https://doi.org/10.1007/s10661-010-1582-5

Cunha, D. G. F., Finkler, N. R., Gómez, N., Cochero, J., Donadelli, J. L., Saltarelli, W. A., … Thomas, S. A. (2019). Agriculture influences ammonium and soluble reactive phosphorus retention in South American headwater streams. *Ecohydrology*, (November), 1–14. https://doi.org/10.1002/eco.2184

Dijkstra &, & Clausnitzer. (2014). *The Dragonflies and Damselflies of Eastern Africa: Handbook for all Odonata from Sudan to Zimbabwe*. (298, Ed.). Studies in Afrotropical Zoology. Retrieved from https://www.africamuseum.be/

Dodds, W. K., Perkin, J. S., & Gerken, J. E. (2013). Human impact on freshwater ecosystem services: A global perspective. *Environmental Science and Technology*, *47*(16), 9061–9068. https://doi.org/10.1021/es4021052

Dolný, A., Bárta, D., Lhota, S., & Drozd, P. (2011). Dragonflies (Odonata) in the Bornean rain forest as indicators of changes in biodiversity resulting from forest modiication and destruction. *Tropical Zoology*, *24*, 63–86. https://doi.org/Article

Ducarme, F., Luque, G. M., & Courchamp, F. (2013). What are “ charismatic species ” for conservation biologists ?, (July), 1–8. Retrieved from https://www.researchgate.net/profile/Frederic_Ducarme/publication/302596828_What_are_charismatic_species_for_conservation_biologists/links/5731e95508ae9ace84047338.pdf

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Naiman, R. J., Knowler, D. J., & Le, C. (2006). Freshwater biodiversity : importance , threats , status and conservation challenges, (2006), 163–182. https://doi.org/10.1017/S1464793105006950

Dutra, S., & Marco, P. De. (2015). Bionomic differences in odonates and their in fl uence on the ef fi ciency of indicator species of environmental quality, *49*, 132–142. https://doi.org/10.1016/j.ecolind.2014.09.016

Ford, R. E. (1990). The dynamics of human-environment interactions in the tropical montane agrosystems of Rwanda: implications for economic development and environmental stability. *Mountain Research and Development*, 43–63. https://doi.org/DOI: 10.2307/3673538

Fu, C., Xu, Y., Bundy, A., Grüss, A., Coll, M., Heymans, J. J., … Shin, Y. (2019). Making ecological indicators management ready : Assessing the specificity , sensitivity , and threshold response of ecological indicators. *Ecological Indicators*, *105*(May), 16–28. https://doi.org/10.1016/j.ecolind.2019.05.055

Gerson Araujo, F., Fichberg, I., Teixeira Pinto, B. C., & Galvao Peixoto, M. (2003). A Preliminary Index of Biotic Integrity for Monitoring the Condition of the Rio Paraiba do Sul, Southeast Brazil. *Environmental Management*. https://doi.org/10.1007/s00267-003-3003-9

Golfieri, B., Hardersen, S., Maiolini, B., & Surian, N. (2016). Odonates as indicators of the ecological integrity of the river corridor: Development and application of the Odonate River Index (ORI) in northern Italy. *Ecological Indicators*, *61*, 234–247. https://doi.org/10.1016/j.ecolind.2015.09.022

Habarurema, E., & Steiner, K. G. (1997). Soil suitability classification by farmers in southern Rwanda. *Geoderma*, *75*(1–2), 75–87. https://doi.org/10.1016/S0016-7061(96)00078-X

Hartter, J., & Ryan, S. J. (2010). Top-down or bottom-up?. Decentralization, natural resource management, and usufruct rights in the forests and wetlands of western Uganda. *Land Use Policy*, *27*(3), 815–826. https://doi.org/10.1016/j.landusepol.2009.11.001

Hope, R. M. (2013). Rmisc: Ryan miscellaneous. R package version 1.5.

Kim, S., & Kim, M. S., (2015). (2015). Package ‘ppcor’. Communications for Statistical Applications and Methods, *22*(6), 665-674.

Kindt, R., Breugel, P. V., Lillesø, J. P. B., Bingham, M., Demissew, S., Dudley, C., … Graudal, L. (2011). *Potential Natural Vegetation of Eastern Africa (Ethiopia , Kenya , Malawi , Rwanda , Tanzania , Uganda and Zambia) VOLUME 5 Description and Tree Species Composition for Other Potential Natural Vegetation Types (Vegetation Types other than Forests , Wood* (Working Pa, Vol. 5). Forest & Landscape Denmark University of Copenhagen 23. Retrieved from www.sl.life.ku.dk

Kipping, J., Günther, A., & Uyizeye, E. (2017). Pseudagrion kamiranzovu sp. No v., a new flagship species of damselfly from Rwanda’s Nyungwe Forest (Odonata: Coenagrionidae). *Odonatologica*, *46*(3–4), 301–318. https://doi.org/10.5281/zenodo.1040318

Lu, Y., Abuel-Naga, H., Leong, E. C., Bouazza, A., & Lock, P. (2018). Effect of water salinity on the water retention curve of geosynthetic clay liners. *Geotextiles and Geomembranes*, *46*(6), 707–714. https://doi.org/10.1016/j.geotexmem.2018.07.002

Luke, S. H., Dow, R. A., Butler, S., Khen, C. V. U. N., Aldridge, D. C., Foster, W. A., … Stretton, A. (2017). The impacts of habitat disturbance on adult and larval dragonflies (Odonata) in rainforest streams in Sabah , Malaysian Borneo. *Freshwater Biology*, 491–506. https://doi.org/10.1111/fwb.12880

Majer, J. D., Callan, S. K., Edwards, K., Gunawardene, N. R., & Taylor, C. K. (2013). Baseline survey of the terrestrial invertebrate fauna of Barrow Island. *Records of the Western Australian Museum Supplement*, (83), 13-112. https://doi.org/10.18195/issn.0313-122x.83.2013.013-112

Maltchik, L., Stenert, C., Kotzian, C. B., Pires, M. M., Maltchik, L., Stenert, C., & Kotzian, C. B. (2010). Responses of Odonate Communities to Environmental Factors in Southern Brazil Wetlands. *Journal of the Kansas Entomological Society*, *83*(3), 208–220. https://doi.org/10.2317/JKES0910.13.1

Mangadze, T., Dalu, T., & William Froneman, P. (2019). Biological monitoring in southern Africa: A review of the current status, challenges and future prospects. *Science of the Total Environment*, *648*, 1492–1499. https://doi.org/10.1016/j.scitotenv.2018.08.252

Markovic, D., Carrizo, S., Freyhof, J., Cid, N., Lengyel, S., Scholz, M., … Darwall, W. (2014). Europe's freshwater biodiversity under climate change: Distribution shifts and conservation needs. *Diversity and Distributions*, *20*(9), 1097–1107. https://doi.org/10.1111/ddi.12232

Marques, M., Eduardo, P., Samuel, P., & Göran, R. (2018). Predicting the effects of future climate change on the distribution of an endemic damselfly (Odonata , Coenagrionidae) in subtropical South American grasslands, 303–319. https://doi.org/10.1007/s10841-018-0063-y

McGeoch, M. A., Sithole, H., Samways, M. J., Simaika, J. P., Pryke, J. S., Picker, M., … Hamer, M. (2011). Conservation and monitoring of invertebrates in terrestrial protected areas. *Koedoe*, *53*(2), 1–13. https://doi.org/10.4102/koedoe.v53i2.1000

McPeek, M. A. (2008). Ecological factors influencing the distribution and abundances of Odonata. *Dragonflies and Damselfiles: Model Organisms for Ecological and Evolutionary Research*, 51–62. Retrieved from https://antioch.on.worldcat.org/oclc/5104977406

Mendes, T. P., Oliveira-Junior, J. M. B., Cabette, H. S. R., Batista, J. D., & Juen, L. (2017). Congruence and the Biomonitoring of Aquatic Ecosystems: Are Odonate Larvae or Adults the Most Effective for the Evaluation of Impacts. *Neotropical Entomology*, *46*(6), 631–641. https://doi.org/10.1007/s13744-017-0503-5

Miguel, T. B., Oliveira-Junior, J. M. B., Ligeiro, R., & Juen, L. (2017). Odonata (Insecta) as a tool for the biomonitoring of environmental quality. *Ecological Indicators*, *81*(June), 555–566. https://doi.org/10.1016/j.ecolind.2017.06.010

Monteiro-Júnior, C. S., Juen, L., & Hamada, N. (2014). Effects of urbanization on stream habitats and associated adult dragonfly and damselfly communities in central Brazilian Amazonia. *Landscape and Urban Planning*, *127*(1), 28–40. https://doi.org/10.1016/j.landurbplan.2014.03.006

Monteiro, S., Juen, L., & Hamada, N. (2015). Analysis of urban impacts on aquatic habitats in the central Amazon basin : Adult odonates as bioindicators of environmental quality. *Ecological Indicators*, *48*, 303–311. https://doi.org/10.1016/j.ecolind.2014.08.021

Mukanyandwi, V., Kurban, A., Hakorimana, E., Nahayo, L., Habiyaremye, G., Gasirabo, A., & Sindikubwabo, T. (2019). Seasonal assessment of drinking water sources in Rwanda using GIS, contamination degree (Cd), and metal index (MI). *Environmental Monitoring and Assessment*, *191*(12). https://doi.org/10.1007/s10661-019-7757-9

Muñoz-Villers, L. E., & López-Blanco, J. (2008). Land use/cover changes using Landsat TM/ETM images in a tropical and biodiverse mountainous area of central-eastern Mexico. *International Journal of Remote Sensing*, *29*(1), 71–93. https://doi.org/10.1080/01431160701280967

Ndayisaba, F., Guo, H., Bao, A., Guo, H., Karamage, F., & Kayiranga, A. (2016). Understanding the spatial temporal vegetation dynamics in Rwanda. *Remote Sensing*, *8*(2), 1–17. https://doi.org/10.3390/rs8020129

Ntwali, D., Ogwang, B. A., & Ongoma, V. (2016). The Impacts of Topography on Spatial and Temporal Rainfall Distribution over Rwanda Based on WRF Model. *Atmospheric and Climate Sciences*, *06*(02), 145–157. https://doi.org/10.4236/acs.2016.62013

Parmar, T. K., Rawtani, D., & Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. *Frontiers in Life Science*, *9*(2), 110–118. https://doi.org/10.1080/21553769.2016.1162753

Paulson, D. (2011). *Dragonflies and Damselflies of the East*. Princeton University Press.

Pereira, D. F. G., de Oliveira Junior, J. M. B., & Juen, L. (2019). Environmental changes promote larger species of Odonata (Insecta) in Amazonian streams. *Ecological Indicators*, *98*(October 2018), 179–192. https://doi.org/10.1016/j.ecolind.2018.09.020

Rabeni, C. F. (2000). Evaluating physical habitat integrity in relation to the biological potential of streams. *Hydrobiologia*, *422–423*, 245–256. https://doi.org/10.1007/978-94-011-4164-2_20

Rebaudo, F., & Dangles, O. (2015). Adaptive management in crop pest control in the face of climate variability: an agent-based modeling approach. *Ecology and Society*, *20*(2), 18. https://doi.org/10.5751/es-07511-200218

Renner, S., Sahlén, G., & Périco, E. (2016). Testing Dragonflies as Species Richness Indicators in a Fragmented Subtropical Atlantic Forest Environment. *Neotropical Entomology*, *45*(3). https://doi.org/10.1007/s13744-015-0355-9

Ripley, B. (2019). MASS: support functions and datasets for Venables and Ripley's MASS. *R Package Version*, *5*(10), 7–3. Retrieved from http://www2.uaem.mx/r-mirror/web/packages/MASS/

Rocha-ortega, M., Rodríguez, P., & Córdoba-aguilar, A. (2019). Can dragon fl y and damsel fl y communities be used as bioindicators of land use intensi fi cation ?, *107*(February). https://doi.org/10.1016/j.ecolind.2019.105553

Romero, E., Gens, A., & Lloret, A. (1999). Water permeability, water retention and microstructure of unsaturated compacted Boom clay. *Engineering Geology*, *54*(1–2), 117–127. https://doi.org/10.1016/S0013-7952(99)00067-8

Salmah, M. R. C., Tribuana, S. W., & Hassan, A. A. (2006). The population of Odonata (dragonflies) in small tropical rivers with reference to asynchronous growth patterns. *Aquatic Insects*, *28*(3), 195–209. https://doi.org/10.1080/01650420600922315

Samways, M, J., & Simaika, J, P. . (2014). Manual of Freshwater Assessment for South Africa: Dradonfly Biotic Index. *Suricata 2*, *1*. https://doi.org/10.1017/CBO9781107415324.004

Samways, M. J. (2008). Dragonflies and damselflies : model organisms for ecological and evolutionary research. *Dragonflies and Damselflies: Model Organisms for Ecological and Evolutionary Research*, *94*(3), 727–728.

Samways, Michael J., & Grant, P. B. C. (2007). Honing Red List Assessments of lesser-known taxa in biodiversity hotspots. *Biodiversity and Conservation*, *16*(9), 2575–2586. https://doi.org/10.1007/s10531-006-9023-z

Sayer, C. A., Máiz-Tomé, L., & Darwall, W. R. T. (2018). *Freshwater biodiversity in the Lake Victoria Basin: Guidance for species conservation, site protection, climate resilience and sustainable livelihoods. International Union for Conservation of Nature*. IUCN Cambridge, UK in collaboration with IUCN Gland, Switzerland. https://doi.org/https://doi.org/10.2305/IUCN.CH.2018.RA.2.en

Schmeller, D. S., Loyau, A., Bao, K., Brack, W., Chatzinotas, A., De Vleeschouwer, F., … Vredenburg, V. T. (2018). People, pollution and pathogens – Global change impacts in mountain freshwater ecosystems. *Science of the Total Environment*, *622–623*, 756–763. https://doi.org/10.1016/j.scitotenv.2017.12.006

Siddig, A. A. H., Ellison, A. M., Ochs, A., Villar-Leeman, C., & Lau, M. K. (2016). How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. *Ecological Indicators*. https://doi.org/10.1016/j.ecolind.2015.06.036

Simaika, J. P., & Samways, M. J. (2009a). An easy-to-use index of ecological integrity for prioritizing freshwater sites and for assessing habitat quality. *Biodiversity and Conservation*, *18*(5), 1171–1185. https://doi.org/10.1007/s10531-008-9484-3

Simaika, J. P., & Samways, M. J. (2009b). Reserve selection using Red Listed taxa in three global biodiversity hotspots: Dragonflies in South Africa. *Biological Conservation*, *142*(3), 638–651. https://doi.org/10.1016/j.biocon.2008.11.012

Simaika, J. P., & Samways, M. J. (2018). Insect conservation psychology. *Journal of Insect Conservation*, *22*(3), 635-642. https://doi.org/10.1007/s10841-018-0047-y

Siziba, N., Matshisela, A., Mwedzi, T., & Bere, T. (2018). Macroinvertebrate communities in riverine systems of buffer areas of protected wildland , rangeland and city areas : implications for conservation of riverine systems on urbanising watersheds. *Environmental Science and Pollution Research*, 758–770. https://doi.org/10.1007/s11356-017-0487-z.

Soesbergen, A. Van, Kimsey, S., & Hill, S. (2019). Potential impacts of agricultural development on freshwater biodiversity in the Lake Victoria basin. *Aquatic Conservation: Marine and Freshwater Ecosystems*, (December 2018), 1052–1062. https://doi.org/10.1002/aqc.3079

Taniwaki, R. H., Piggott, J. J., Ferraz, S. F. B., & Matthaei, C. D. (2017). Climate change and multiple stressors in small tropical streams. *Hydrobiologia*, *793*(1), 41–53. https://doi.org/10.1007/s10750-016-2907-3

Turak, E., Harrison, I., Dudgeon, D., Abell, R., Bush, A., Darwall, W., … De Wever, A. (2017). Essential Biodiversity Variables for measuring change in global freshwater biodiversity. *Biological Conservation*, *213*, 272–279. https://doi.org/10.1016/j.biocon.2016.09.005

Vaccaro, I., Beltran, O., Paquet, P. A., Barcelona, U. De, Building, S. L., & Barcelona, U. De. (2012). Political ecology and conservation policies : some theoretical genealogies. *Journal of Political Ecology*, *20*(1), 255-272.

Valente-Neto, F., De Oliveira Roque, F., Rodrigues, M. E., Juen, L., & Swan, C. M. (2016). Toward a practical use of Neotropical odonates as bioindicators: Testing congruence across taxonomic resolution and life stages. *Ecological Indicators*, *61*, 952–959. https://doi.org/10.1016/j.ecolind.2015.10.052

Villéger, S. N. W. H. M. (2008). New multidimensional functional diversity indices for a multifaceted framework in functional ecology. *Ecological Society of America*, *89*, *Issue*(8), 2290–2301. https://doi.org/https://doi.org/10.1890/07-1206.1

Vorster, C., Samways, M. J., Simaika, J. P., Kipping, J., Clausnitzer, V., Suhling, F., & Dijkstra, K. B. (2020). Development of a new continental-scale index for freshwater assessment based on dragonfly assemblages. *Ecological Indicators*, *109*(April 2019), 105819. https://doi.org/10.1016/j.ecolind.2019.105819

Walsh, C. J., Waller, K. A., Gehling, J., & Mac Nally, R. (2007). Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. *Freshwater Biology*, *52*(3), 574–587. https://doi.org/10.1111/j.1365-2427.2006.01706.x

Wickham, H. (2016). (2016). *ggplot2: elegant graphics for data analysis.* Springer

Appendices

Appendix 1-3: DBI scores for each species. DBI consists of Distribution-Based Score (DBS), Sensitivity-Based Score (SBS) and Threat-Based Score (TBS) for each species recorded during the survey of 2019. The meaning of each score can be found in Table 2-2

	Species	DBS	SBS	TBS
1	*Acisoma trifidum*	1	1	0
2	*Acisoma variegatum*	1	1	0
3	*Aethriamanta rezia*	3	2	0
4	*Africallagma elongatum*	1	1	0
5	*Africallagma pseudelongatum*	2	1	0
6	*Africallagma vaginale*	3	3	0
7	*Afroaeschna scotias*	3	3	0
8	*Agriocnemis forcipata*	3	3	1
9	*Agriocnemis gratiosa*	1	1	0
10	*Agriocnemis inversa*	1	1	0
11	*Agriocnemis palaeforma*	3	3	3
12	*Agriocnemis victoria*	2	1	0
13	*Anaciaeschna triangulifera*	3	1	0
14	*Anax imperator*	0	1	0
15	*Anax speratus*	3	2	0
16	*Anax tristis*	3	2	0
17	*Atoconeura eudoxia*	3	3	0
18	*Atoconeura pseudeudoxia*	3	3	0
19	*Brachythemis leucosticta*	0	1	0
20	*Ceriagrion glabrum*	1	1	0
21	*Ceriagrion platystigma*	2	2	0
22	*Chalcostephia flavifrons*	3	2	0
23	*Crocothemis erythraea*	1	1	0
24	*Crocothemis sanguinolenta*	3	1	0
25	*Diplacodes lefebvrii*	1	1	0
26	*Diplacodes luminans*	1	1	0
27	*Diplacodes pumila*	3	3	0
28	*Hemistigma albipunctum*	1	1	0
29	*Ictinogomphus ferox*	1	1	0
30	*Ischnura senegalensis*	1	0	0
31	*Lestes dissimulans*	3	3	0
32	*Lestes virgatus*	3	3	0
33	*Neodythemis nyungwe*	3	3	3
34	*Nesciothemis farinosa*	1	1	0
35	*Notogomphus lujai*	3	3	0
36	*Olpograstra ingubis*	3	2	0

	Species	DBS	SBS	TBS
37	*Orthetrum abbotti*	3	1	0
38	*Orthetrum austeni*	3	1	0
39	*Orthetrum brachiale*	0	0	0
40	*Orthetrum caffrum*	2	1	0
41	*Orthetrum camerunense*	1	1	0
42	*Orthetrum chrysostigma*	1	0	0
43	*Orthetrum guineense*	3	0	0
44	*Orthetrum hintzi*	3	2	0
45	*Orthetrum julia*	2	0	0
46	*Orthetrum microstigma*	3	2	0
47	*Orthetrum stemmale*	1	2	0
48	*Orthetrum trinacria*	3	2	0
49	*Othetrum chrysostigma*	1	1	0
50	*Palpopleura deceptor*	3	3	0
51	*Palpopleura jucunda*	3	2	0
52	*Palpopleura lucia*	0	0	0
53	*Palpopleura portia*	0	0	0
54	*Pantala flavescens*	0	0	0
55	*Paragomphus genei*	3	1	0
56	*Parazyxomma flavicans*	3	3	3
57	*Phaon iridipennis*	2	0	0
58	*Phyllomacromia contumax*	3	3	0
59	*Platycypha caligata*	2	2	0
60	*Proischnura subfurcata*	0	0	0
61	*Pseudagrion hageni*	3	0	1
62	*Pseudagrion hamoni*	1	0	0
63	*Pseudagrion kamiranzovu*	3	3	3
64	*Pseudagrion kersteni*	1	0	0
65	*Pseudagrion massaicum*	1	0	0
66	*Pseudagrion nubicum*	1	1	0
67	*Pseudagrion sjoestedti*	3	2	0
68	*Pseudagrion spernatum*	1	0	0
69	*Pseudagrion sublacteum*	1	1	0
70	*Rhyothemis fenestrina*	3	3	0
71	*Rhyothemis semihyalina*	3	2	0

	Species	DBS	SBS	TBS
72	*Stenocypha jacksoni*	3	2	1
73	*Stenocypha tenuis*	3	2	0
74	*Sympetrum fonscolombii*	3	0	0
75	*Tholymis tillarga*	2	1	0
76	*Tramea basilaris*	3	2	0
77	*Tretrathemis camerunensis*	3	3	0
78	*Trithemis annulata*	1	1	0
79	*Trithemis arteriosa*	1	1	0
80	*Trithemis dorsalis*	3	1	0
81	*Trithemis hecate*	3	1	0
82	*Trithemis nuptialis*	2	1	0
83	*Trithemis pluvialis*	2	1	0
84	*Trithemis stictica*	2	1	0
85	*Trithemis werneri*	3	1	0
86	*Trithetrum navasi*	3	2	0
87	*Urothemis assignata*	1	2	0
88	*Urothemis edwardsii*	2	2	0
89	*Zosteraeschna ellioti*	1	2	0
90	*Zygonyx natalensis*	3	0	0
91	*Zygonyx torridus*	3	0	0

Appendix 2-3: Unique species per locality and ecological zone

Ecological Zone	Locality	Unique Species
North East	Akagera National Park	*Lestes dissimulans* Fraser, 1955
		Africallagma vaginale Sjöstedt, 1917
		Agriocnemis palaeforma Pinhey, 1959
		Ceriagrion platystigma Fraser, 1941
		Anaciaeschna triangulifera McLachlan, 1896
		Phyllomacromia contumax Selys, 1879
		Orthetrum trinacria Selys, 1841
		Palpopleura deceptor Calvert, 1899
		Parazyxomma flavicans Martin, 1908
		Tetrathemis camerunensis Sjöstedt, 1900
		Trithetrum navasi Lacroix, 1921
South East	Jarama wetland	*Orthetrum machadoi** Longfield, 1955
South Central	Rugende wetland	*Olpogastra lugubris** Karsch, 1895
	Buzana wetland	*Orthetrum abbotti** Calvert, 1892
North West	Giswati National Park	*Notogomphus lujai* Schouteden, 1934
		*Sympetrum fonscolombii** Selys, 1840
North Central	Rugezi wetland	*Diplacodes pumila** Dijkstra, 2006
South West	Nyungwe National Park	*Stenocypha jacksoni** Pinhey, 1952
		*Pseudagrion kamiranzovu*** Kipping et al., 2017
		Atoconeura pseudeudoxia Longfield, 1953
		Neodythemis Nyungwe Dijkstra & Vick, 2006
	Cyamudondo, Nyungwe National Park	*Stenocypha tenuis** Longfiled 1936
		Atoconeura pseudeudoxia Longfield, 1953
		Orthetrum hintzi Schmidt, 1951
		*Trithemis dorsalis** Rambur, 1842
	Farmakina, Kamembe	*Agriocnemis forcipata* Le Roi, 1915
		*Orthetrum austeni** Kirby, 1900
		Orthetrum hintzi Schmidt, 1951
	Ruhwa river	*Trithemis werneri** Ris, 1912

Species with (**) are globally endemic to the mentioned habitats, while those with (*) are nationally unique to their habitats.

Appendix 3-3: Checklist for odonate species for Rwanda

(black)	Sub-orders and Families
(grey)	Genus
(white)	Species
	Each of these has the name of the person who described it and the year it was proven new to science

	Check-list	**From this study**	**North East**	**South East**	**North Cent-ral**	**South Cent-ral**	**North West**	**South West**
ZYGOPTERA Selys, 1854								
Lestidae Calvert, 1901								
***Lestes* Leach, 1815**								
***Lestes* Leach, 1815 *virgatus*-group = Africalestes Kennedy, 1920**								
Lestes virgatus Burmeister, 1839	x							
***Lestes* Leach, 1815 *tridens*-group = Paralestes Schmidt, 1951 s.s.**								
Lestes dissimulans Fraser, 1955	x	x	x					
Calopterygidae Selys, 1850								
***Phaon* Selys, 1853**								
Phaon camerunensis Sjöstedt, 1900	x							
Phaon iridipennis Burmeister, 1839	x	x			x	x		
***Umma* Kirby, 1890**								
Umma saphirina Förster, 1916	x							
Chlorocyphidae Cowley, 1937								
***Chlorocypha* Fraser, 1928**								
Chlorocypha flammea Dijkstra & Clausnitzer, 2015	x							
***Platycypha* Fraser, 1949**								
Platycypha caligata Selys, 1853	x	x				x		x
***Stenocypha* Dijkstra, 2013**								
Stenocypha jacksoni Pinhey, 1952	x	x						x
Stenocypha tenuis Longfiled 1936								x
***Allocnemis* Selys, 1863**								
Allocnemis pauli Longfield, 1936	x							
Coenagrionidae Kirby, 1890								

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
***Aciagrion Selys*, 1891**								
Aciagrion heterostictum Fraser, 1955	x							
***Africallagma* Kennedy, 1920**								
Africallagma elongatum Martin, 1907	x	x	x	x		x	x	
Africallagma pseudelongatum Longfield, 1936	x						x	x
Africallagma vaginale Sjöstedt, 1917	x	x	x					
***Agriocnemis* Selys, 1877**								
Agriocnemis forcipata Le Roi, 1915	x	x						x
Agriocnemis gratiosa Gerstäcker, 1891	x	x	x	x		x	x	x
Agriocnemis inversa Karsch, 1899	x	x	x	x		x	x	
Agriocnemis palaeforma Pinhey, 1959	x	x	x	x				
Agriocnemis victoria Fraser, 1928	x	x		x		x	x	
***Azuragrion* May, 2002**								
nigridorsum Selys, 1876	x							
***Ceriagrion* Selys, 1876**								
***Ceriagrion* Selys, 1876 glabrum-group**								
Ceriagrion corallinum Campion, 1914	x							
Ceriagrion glabrum Burmeister, 1839	x	x	x	x	x	x	x	x
***Ceriagrion* Selys, 1876 varians-group**								
Ceriagrion platystigma Fraser, 1941	x	x	x					
***Ischnura* Charpentier, 1840**								
Ischnura senegalensis Rambur, 1842	x	x		x	x	x	x	x
***Proischnura* Kennedy, 1920**								
Proischnura subfurcata Selys, 1876	x	x	x	x	x	x	x	x
***Pseudagrion* Selys, 1876**								
***Pseudagrion* Selys, 1876 A–group**								

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
Pseudagrion A *hageni* Karsch, 1893	x	x				x		x
Pseudagrion A *kamiranzovu* Kipping, Günther & Uyizeye, 2017	x							x
Pseudagrion A *kersteni* Gerstäcker, 1869	x	x	x		x	x	x	x
Pseudagrion A *spernatum* Selys, 1881	x	x				x	x	x
Pseudagrion Selys, 1876 B–group								
Pseudagrion B *glaucescens* Selys, 1876	x							
Pseudagrion B *hamoni* Fraser, 1955	x	x	x	x	x	x		x
Pseudagrion B *isidromorai* Compte Sart, 1967	x							
Pseudagrion B *massaicum* Sjöstedt, 1909	x	x	x	x			x	x
Pseudagrion B *nubicum* Selys, 1876	x	x	x	x	x	x	x	
Pseudagrion B *sjoestedti* Förster, 1906	x	x		x		x		
Pseudagrion B *sublacteum* Karsch, 1893	x	x	x	x	x			x
ANISOPTERA Selys, 1854								
Aeshnidae Leach, 1815								
***Afroaeschna* Peters & Theischinger, 2011**								
Afroaeschna scotias Pinhey, 1952	x	x						x
***Anaciaeschna* Selys, 1878**								
Anaciaeschna triangulifera McLachlan, 1896	x	x	x					
***Anax* Leach, 1815**								
Anax ephippiger Burmeister, 1839	x							
Anax imperator Leach, 1815	x	x	x	x	x	x	x	x
Anax speratus Hagen, 1867	x	x	x			x		
Anax tristis Hagen, 1867	x	x	x					x
***Gynacantha* Rambur, 1842**								
***Gynacanth*a Rambur, 1842 africana-group**								

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
Gynacantha A *villosa* Grünberg, 1902	x							
***Zosteraeschna* Peters & Theischinger, 2011**								
Zosteraeschna ellioti Kirby, 1896	x	x	x	x		x	x	x
Gomphidae Rambur, 1842								
***Crenigomphus* Selys, 1892**								
Crenigomphus hartmanni Förster, 1898	x							
***Ictinogomphus* Cowley, 1934**								
Ictinogomphus ferox Rambur, 1842	x	x	x		x	x		x
***Microgomphus* Selys, 1858**								
Microgomphus nyassicus Grünberg, 1902	x							
ANISOPTERA Selys, 1854								
***Notogomphus* Selys, 1858**								
Notogomphus flavifrons Fraser, 1952	x							
Notogomphus gorilla Dijkstra, 2015	x							
Notogomphus lujai Schouteden, 1934	x	x					x	
Paragomphus Cowley, 1934								
Paragomphus genei Selys, 1841	x	x				x		x
Libelluloidea *incertae sedis*								
Macromiidae Needham, 1903								
***Phyllomacromia* Selys, 1878**								
Phyllomacromia contumax Selys, 1879	x	x	x					
Phyllomacromia picta Hagen in Selys, 1871	x							
Libellulidae Leach, 1815								
***Acisoma* Rambur, 1842**								
Acisoma trifidum Kirby, 1889	x	x	x	x	x	x	x	x
Acisoma variegatum Kirby, 1898	x	x	x	x			x	x
***Aethriamanta* Kirby, 1889**								

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
Aethriamanta rezia Kirby, 1889	x	x	x	x				
***Atoconeura* Karsch, 1899**								
Atoconeura eudoxia Kirby, 1909	x	x					x	x
Atoconeura pseudeudoxia Longfield, 1953	x	x						x
***Brachythemis* Brauer, 1868**								
Brachythemis impartita Karsch, 1890	x							
Brachythemis leucosticta Burmeister, 1839	x	x	x	x	x	x	x	x
Chalcostephia Kirby, 1889								
Chalcostephia flavifrons Kirby, 1889	x	x	x		x			
***Crocothemis* Brauer, 1868**								
Crocothemis erythraea Brullé, 1832	x	x	x	x	x	x	x	x
Crocothemis sanguinolenta Burmeister, 1839	x	x	x			x		
***Diplacodes* Kirby, 1889**								
Diplacodes lefebvrii Rambur, 1842	x	x		x	x	x	x	x
Diplacodes luminans Karsch, 1893	x	x	x		x	x	x	x
Diplacodes pumila Dijkstra, 2006	x	x			x			
***Hadrothemis* Karsch, 1891**								
Hadrothemis versuta Karsch, 1891	x							
***Hemistigma* Kirby, 1889**								
Hemistigma albipunctum Rambur, 1842	x	x	x	x		x	x	x
***Neodythemis* Karsch, 1889**								
Neodythemis nyungwe Dijkstra & Vick, 2006	x							x
***Nesciothemis* Longfield, 1955**								
***Nesciothemis* Longfield, 1955**								
Nesciothemis farinosa Förster, 1898	x	x	x	x	x	x	x	x
***Notiothemis* Ris, 1919**								
Notiothemis jonesi Ris, 1919	x							

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
***Olpogastra* Karsch, 1895**								
Olpogastra lugubris Karsch, 1895	x	x				x		
***Orthetrum* Newman, 1833**								
Orthetrum abbotti Calvert, 1892	x	x				x		
Orthetrum austeni Kirby, 1900	x	x						x
Orthetrum brachiale Palisot de Beauvois, 1817	x	x	x	x		x	x	x
Orthetrum caffrum Burmeister, 1839	x	x				x	x	x
Orthetrum camerunense Gambles, 1959	x	x	x			x	x	x
Orthetrum chrysostigma Burmeister, 1839	x	x	x			x	x	x
Orthetrum guineense Ris, 1910	x	x		x		x		
Orthetrum hintzi Schmidt, 1951	x	x						x
Orthetrum julia Kirby, 1900	x	x	x			x		x
Orthetrum machadoi Longfield, 1955	x	x		x				
Orthetrum microstigma Ris, 1911	x	x				x		x
Orthetrum stemmale Burmeister, 1839	x	x	x	x	x	x		
Orthetrum trinacria Selys, 1841	x	x	x					
***Palpopleura* Rambur, 1842**								
Palpopleura deceptor Calvert, 1899	x	x	x					
Palpopleura jucunda Rambur, 1842	x	x		x		x		
Palpopleura lucia Drury, 1773	x	x	x	x		x	x	x
Palpopleura portia Drury, 1773	x	x	x	x		x	x	x
***Pantala* Hagen, 1861**								
Pantala flavescens Fabricius, 1798	x	x	x	x		x	x	x
***Parazyxomma* Pinhey, 1961**								
Parazyxomma flavicans Martin, 1908	x	x	x					
***Rhyothemis* Hagen, 1867**								
Rhyothemis fenestrina Rambur, 1842	x	x	x	x				

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
Rhyothemis notata Fabricius, 1781	x							
Rhyothemis semihyalina Desjardins, 1832	x	x	x					x
***Sympetrum* Newman, 1833**								
Sympetrum fonscolombii Selys, 1840	x	x					x	
***Tetrathemis* Brauer, 1868**								
Tetrathemis camerunensis Sjöstedt, 1900	x	x	x					
***Tholymis* Hagen, 1867**								
Tholymis tillarga Fabricius, 1798	x	x	x	x			x	
***Tramea* Hagen, 1861**								
Tramea basilaris Palisot de Beauvois, 1817	x	x	x			x		
***Trithemis* Brauer, 1868**								
***Trithemis* Brauer, 1868 annulata-group**								
Trithemis annulata Palisot de Beauvois, 1807	x	x	x		x	x	x	
Trithemis arteriosa Burmeister, 1839	x	x	x	x	x	x	x	x
***Trithemis* Brauer, 1868 basitincta-group**								
Trithemis donaldsoni Calvert, 1899	x							
***Trithemis* Brauer, 1868 dorsalis-group**								
Trithemis dichroa Karsch, 1893	x							
Trithemis dorsalis Rambur, 1842	x	x						x
Trithemis pluvialis Förster, 1906	x	x	x		x	x		
***Trithemis* Brauer, 1868 stictica-group**								
Trithemis nuptialis Karsch, 1894	x	x	x	x		x		
Trithemis stictica Burmeister, 1839	x	x	x	x		x		
***Trithemis* Brauer, 1868 monotypic groups**								
Trithemis hecate Ris, 1912	x	x		x		x		
Trithemis werneri Ris, 1912	x	x						x

	Check-list	From this study	North East	South East	North Cent-ral	South Cent-ral	North West	South West
***Trithetrum* Dijkstra & Pilgrim, 2007**								
Trithetrum navasi Lacroix, 1921	x	x	x					
Urothemis Brauer, 1868	x							
Urothemis assignata Selys, 1872	x	x	x	x	x	x	x	
Urothemis edwardsii Selys, 1849	x	x	x		x	x	x	
***Zygonyx* Hagen, 1867**								
Zygonyx natalensis Martin, 1900	x	x	x		x			
Zygonyx regisalberti Schouteden, 1934	x							
Zygonyx torridus Kirby, 1889	x	x	x			x		

Chapter 4: The Application of Odonates as Indicators for Monitoring Freshwater Habitat Integrity

Abstract

Freshwaters are essential habitats to many organisms and suppliers of vital ecosystem services, however, they are increasingly under threats from human practices such as agriculture and mining. It is therefore vital that integrity of these habitats is monitored. Odonates (dragonflies), insects that are highly sensitive to environmental degradation and pollution, could serve as valuable indicators of habitat degradation and integrity. This study evaluates the potential use of odonates in monitoring freshwater habitats by assessing the impact of mining and agriculture on freshwater habitats in Rwanda through the use of the Dragonfly Biotic Index (DBI), individual indicator species, and comparisons using environmental and physical-chemical characteristics. I compared agricultural and mining sites with their reference sites. Additionally, I predict the occurrence of the most abundant odonate species using physical-chemical collected from the field, bioclimatic and hydrological variables from open source databases. Overall, results showed that the DBI of agricultural and mining sites are slightly different, however, the significant differences in DBI was between each of the land use and it reference sites, which suggests the relationship of the two land uses and negative changes in ecological conditions. This is also reflected in changes of habitat characteristics. Riparian vegetation was significantly affected by both practices. Additionally, agriculture was associated with higher electric conductivity in water and slightly higher water temperature. Mining was strongly associated with water turbidity and more sandy substrates. The bioclimatic and hydrological variables that most influence occurrence of odonates are precipitation of the coldest quarter, conditioned elevation and flow accumulation. Ecological friendly land use practices and the restoration of degraded habitats, particularly in riparian

zones, may help mitigate the impacts of detrimental human activities and climate change. My results highlight the effectiveness using odonate-based indices in monitoring ecosystems, and the use of odonate in monitoring can potentially steer sustainable and more environmentally friendly practices while preserving the integrity of freshwater ecosystems.

Key words: odonates, dragonflies, damselflies, mining, agriculture, habitat integrity, wetlands, freshwater ecosystems.

Introduction

Human activities such as agriculture and mining are threats to freshwater ecosystems (Mugni et al., 2013; Dedieu et al., 2015; Wurtsbaugh et al., 2019). In many African countries intensive agriculture in wetlands is encouraged to improve the national economy alleviate poverty (Butchart et al., 2018; Nsengimana et al., 2017). Mining contributes a lot to national revenue and is an employment opportunity to many people (Hilson, 2002; Maconachie et al., 2019). However, these practices compromise the ecological integrity of wetlands by producing pollutants that severely affect biodiversity in freshwater ecosystems. Not only can these pollutants be detrimental to ecosystem functioning, but they also impair ecosystem services, like water for drinking, food, manufacturing irrigation, recreation and navigation, regardless of the scale of these practices (Cunha et al., 2019; Khan et al., 2019; Monteiro-Júnior et al., 2014).

Be it large or small scale, both agriculture and mining considerably impact freshwater ecosystems, especially when operated within close proximity to these ecosystems (Rothenberg et al., 2014; Sievers et al., 2018). Intensified agriculture is also an issue for wetlands in tropical developing countries where rice is one of the main, intensively grown crops (Uwimana et al., 2018; Rothenberg et al., 2014). Growing rice often involves the application of pesticides and fertilizers, which are major pollutants to wetlands (Cunha et al., 2019; Wurtsbaugh et al., 2019). Small scale mining, mostly in the form of artisanal mines, is the most prevalent type of mining in developing countries. Mines are mostly located in alluvial areas, common within and around wetland ecosystems, which poses a direct threat to the integrity of freshwater ecosystems (Dedieu et al., 2015). These practices involve digging soil out of water bodies, which degrades riparian zone and gives rise to a series of other ecological issues (García-García et al., 2017; Salmah et al., 2006).

In addition to the similarities that agriculture and mining display in regards to their impact on freshwater ecosystems, there are several other reasons that make the study of the impacts of these two land use types on wetlands worth exploring together. Both can be assessed through a socio-economic lens. For example, general population trends show that human density is higher in regions where there is fertile arable soils, good for agriculture, and similarly for areas rich in mineral resources (Hilson, 2002). Additionally, the two land uses sustain each in a variety of ways. For example, in east Africa, artisanal mining can be used as a short-term activity that is utilized transitionally on the way to farming, and vice versa (Jønsson & Bryceson, 2009). Often the gain from mining is invested into long-term farming as a result of market instability that most minerals experience (Jønsson & Bryceson, 2009; Patz et al., 2004).

It is important to be grounded in a good understanding of the broader interdisciplinary context of these practices prior to encouraging alternative practices that are environmentally friendlier. In tropical African countries, there is a tremendously high demand for food and resources for infrastructure (Imasiku et al., 2020; Somma, 2015). Rwanda, in particular, is the most densely populated country on the continent, and also has the fastest growing economy in east African (Imasiku et al., 2020). Unfortunately, these development and food demands are accompanied with compromises to freshwater ecosystems. For instance, the rapid pace of infrastructure development has driven over-extraction of sand, rocks and gravel in rivers and streams (Dusková & Machácek, 2013). Agricultural intensification has increasingly put pressure on unprotected wetlands, given that they are the only remaining undeveloped arable areas in the country (Salmah et al., 2006; Uwimana et al., 2018). These alterations to wetlands reduce functional resilience to threats such as effects of climate change, which include severe storms and flooding that have become more frequent in East Africa (Wassila et al., 2018).

To promote environmentally friendly practices and remediation, ecological feedback as responses to agriculture and mining industries should be monitored (Mangadze et al, 2019; Peyre et al., 2001). The effects on ecosystems can be reflected in indicator species' assemblages. Here, odonate species' assemblages is referred to as composition of odonate species in a habitat (Stewart & Samways, 1998). According to ecological niche and bioindication theory, some organisms have specific positions and functions within their habitat (Khatibi & Sheikholeslami, 2016). For example, odonates play multiple roles within the ecosystems they inhabit, serving as voracious predators, but also as prey to a variety of organism, which influences energy cycling and transforming (Miguel et al., 2017; Siddig et al., 2016; Vanacker et al., 2018). A negative change in the richness or abundance (or complete loss) of odonates may impact the entire habitat and reflect the extent to which the whole habitat is degraded or polluted and predicts effects occurring at the ecosystem-level (Remsburg & Turner, 2009; Clausnitzer et al., 2009). Thus, in freshwater ecosystems, indicator species, such as odonate assemblages, will show high sensitivity to changes in physical and chemical parameters. Regular monitoring will thus detect changes in ecosystems and this information can be used to consistently inform decision making.

The primary objective of this study was to evaluate the potential use of odonate-based indicators in habitat integrity monitoring. These consist of the Dragonfly Biotic Index (DBI) developed by Uyizeye et al. (In prep) and individual indicator species indicator as generated by Indicator Value Function of R software (R Core Team 2020) in conjunction with habitat integrity categories (Dufrene & Legendre, 1997; Dutra & Marco, 2015). I explored how agriculture and mining are associated with freshwater habitat integrity, focusing on the potential impact of wetland rice cropping and alluvial open pit mining to streams, rivers and wetlands in Rwanda using odonates as indicators. I also explore bioclimatic (gridded temperature and precipitation) variables from WORLDCLIM (Fick & Hijmans, 2017; Waltari et al.,2014) and hydrological variables that are most closely associated with the occurrence of odonates, focusing on the most abundant species. The anticipation is that odonates can offer an opportunity to monitor how habitat integrity changes as a function of agriculture and mining in a changing climate.

Methods

Study area and sites

The study was conducted in fourteen open pit mining sites (cassiterites, colta and sand mining) in western Rwanda as well as nine rice paddies located in east-central Rwanda (Figure 1-2 & 2-2). Six reference sites for mining were selected in Nyungwe National Park about 40 km north of mining sites. These reference sites were located within the elevation range of the selected mining sites (1800-2000m). Seven upstream sites (at least 100 m away) from the selected mining sites with minimal impacts evident were selected as additional reference sites for mining. Seven reference sites for the agricultural sites (rice paddy) were selected in Akagera National Park within the elevation range of the selected agricultural sites (1287-1306 m). Reference sites were used for comparisons to understand the extent to which agriculture and mining may influence habitat quality (Figure 1-4). While “reference site” can be used interchangeably with “control site”, in this particular study, I use “reference” in order to acknowledge limitations in knowledge of other contributing factors affecting the selected reference sites.

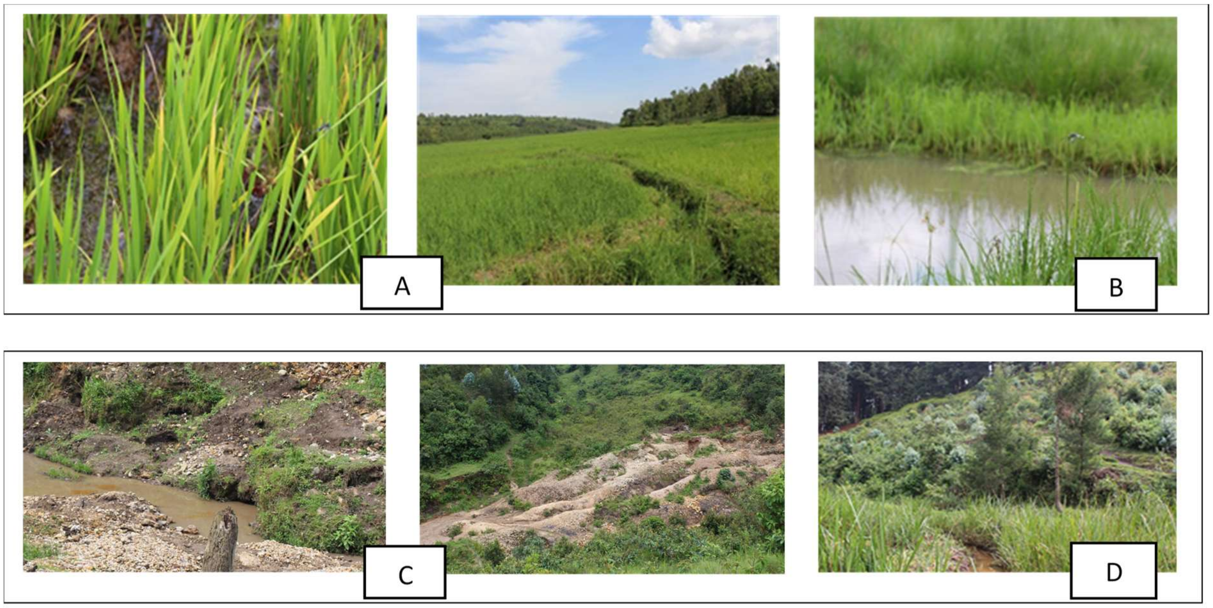

Figure 1-4. Examples of agricultural and mining sites: (A) rice paddy site; (B) Reference sites for agricultural selected in Akagera National Park; (C) mining site; and (D) mining reference site. Photo credit: Erasme Uyizeye

Sampling

In addition to agricultural and mining sites, I sampled 99 sites distributed across six ecological zones, which also include reference sites of mining and agriculture, see chapter 3 (Uyizeye et al. n.d.). The sites in ecological zones and in the two land use types were visited during the short rainy and dry seasons of 2019. These sample sites provided information about with patterns of species distribution and status of habitat integrity throughout the country. Habitat integrity can be defined as the effectiveness in supporting geomorphology, hydrology and ecology of a habitat (Caniani et al., 2016; Gerson et al., 2003). These can be reflected in both odonate assemblages (Miguel et al., 2017) and habitat characteristics such as water body substrates, macrophytes and riparian vegetation (Caniani et al., 2016; Monteiro-Júnior et al., 2014). The surveyed sites are labeled as "Sites per Ecological Zone" in Figure 2-4. At each sample site, I sampled adult odonates, physical-chemical variables, as well as habitat characteristics consisting of water body substrates, macrophytes and riparian vegetation.

Sampling odonates.

To collect species, I used a combination of observations at a distance with necked eyes or binoculars at distance when details cannot be observed by necked eyes. I used direct catch sampling with a sweep net and most of records were identified in the field following the field handbook of Dijkstra and Clausnitzer (2014). At each of the 99 sites sampled, odonate adults were collected for one hour when just myself was sampling, or 0.5 hours when myself and a field assistant were sampling. To ensure that all the present species are active, sampling was conducted between 09 am and 05 pm, only when the weather was sunny and wind was at a minimum (wind speed $\leq$8 km/h) with temperatures above 19° C; odonates tend to decrease their activity below this temperature (Dutra & Marco, 2015). Adults of odonates were collected or observed along a reach of 100 m. A sample point that looks most representative of the whole habitat was identified in the middle of the 100 m stretch to measure water physical-chemical and habitat characteristics (Walsh et al., 2007). By walking back and forth along one bank of the water channel, any species observed within 10 m perpendicular to the water body was caught if possible, identified and kept in paper envelopes. Adults were collected using a sweep net. Species recorded were either flying or perching in the middle or above water body or riparian zone. Caught specimens were washed in acetone and dried after every day of field work before putting them into the envelopes. In addition to odonate sampling, habitat characteristics were recorded as well as GPS coordinates for sample site using WGS_1984 datum.

Sampling physical-chemical variables. I sampled water pH by Oakton pH meter, dissolved oxygen (DO) by Extech Dissolved Oxygen Meter, turbidity by secchi turbidity tube (with the range of 0 to 60 cm), water temperature and electrical conductivity using an Oakton Conductivity Meter, and air temperature, air humidity and wind speed using a Kestrel 4000 Weather Meter. All these variables were measured at the sample point located in the middle of 100 m stretch along each point. Adult odonates were sampled walking back and forth along the 100 m and double count of individuals was avoided as much as possible (Golfieri et al., 2016; Jorge et al., 2011; Tichanek & Tropek, 2016). After odonate sampling was completed, water samples were collected for measuring nutrient (nitrites and phosphates) concentrations. For phosphates, I used a Checker Phosphate Calorimenter with range of 0-2.25 ppm. For nitrites, I used Checker Nitrite Calorimenter with a range of 0-200 ppb. This calorimeter has a small range because nitrite has very low concentrations in water systems (García-García et al., 2017). I chose to measure nitrite over other forms nitrogen because of its higher toxicity to aquatic life in contrast to nitrates, for example. Where needed, it is possible to convert measured nitrites into estimated concertation of other nitrogen forms, such as nitrates and ammonia (Cunha et al., 2019; Voß & Schäfer, 2017; Wurtsbaugh et al., 2019). Additionally, GPS coordinates were recorded at each sample site, to generate a map of species abundance in all sites.

Determining habitat characteristics. At each sample point in the middle of the 100 m stretch, I assessed structures and substrates at the bottom surface of the water bodies sampled. I defined substrate types based on their sizes and I estimated their percentage in relation to other substrates. These included sands: particle size <2mm, gravel: 2–25mm, and rocks: >25mm, as well as silt deposit, riparian vegetation and % canopy cover above a studied water body.

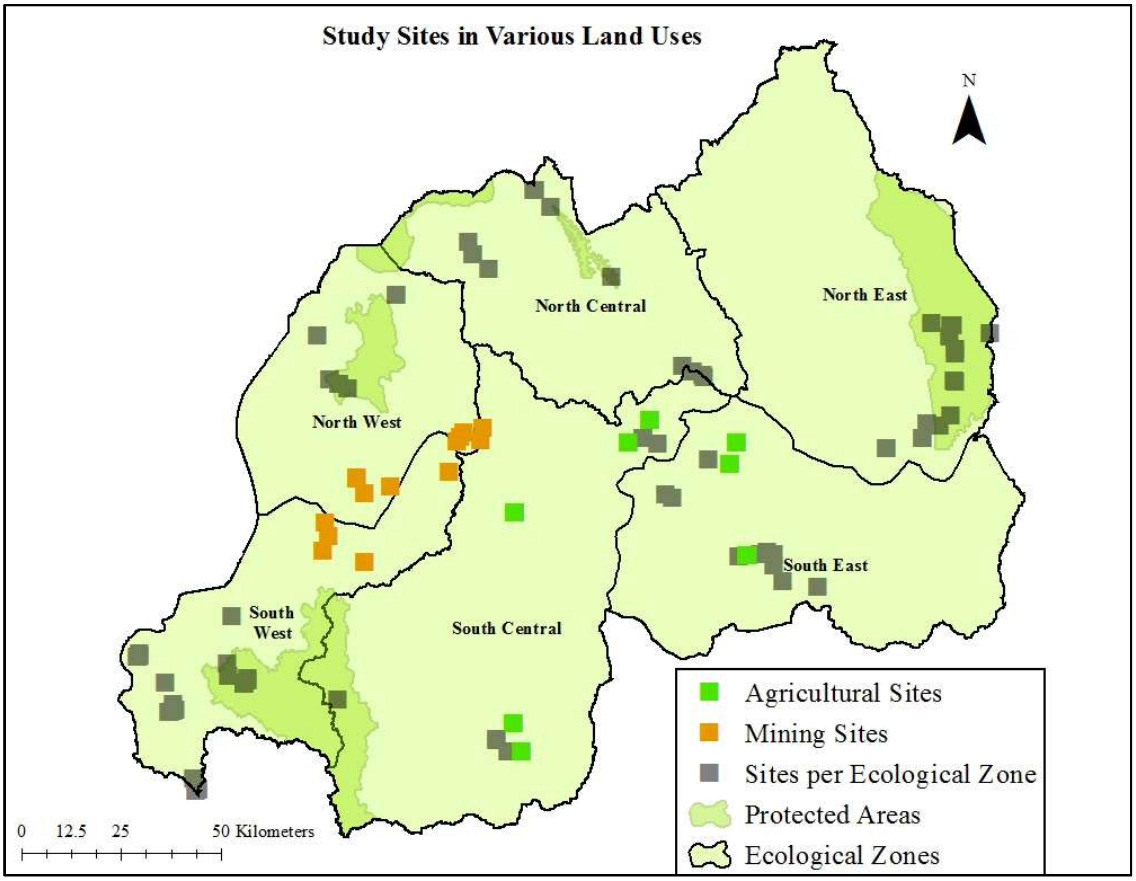

Figure 2-4. Map of wetland study sites across the ecological zones of Rwanda. Sites are identified by their location in agricultural (green), mining (orange), or sites per ecological zone (grey) areas. Sites Ecological Zone are sites that were systematically surveyed per ecological zone to understand species distribution across the country (Uyizeye et al., in prep.).

Analysis

Habitat conditions and integrity: To determine ecological conditions at each site of the six ecological zones, I calculated the Dragonfly Biotic Index (DBI) at each site (Uyizeye et al., in prep.). The DBI score for each recorded species is the sum of each species' Distribution-Based Scores (DBS), Threat Based-Scores (TBS) and Sensitivity-Based Scores (SBS): "$DBI_1+DBI_2+DBI_3+\ldots DBI_N$" (Samways & Simaika, 2014; Simaika & Samways, 2009; Uyizeye et al., in prep; Vorster et al., 2020). A sum of DBI scores of all species and divided by the total number of species recorded at a site provided a score used to compared sites. Higher DBI indicates a healthier or more intact ecological system.

I calculated the Habitat Integrity Index (HII), following the approach used by Luke et al. (2017) and Monteiro-Júnior et al. (2014) HII consists of environmental conditions of water bodies reflected in quality of riparian zone, types of land use and potential sources of pollutants (Luke et al., 2017; Monteiro-Júnior et al., 2014). Scores of HII range from poor quality (0) to good quality (1), the following categories were identified based on the HII score for each site: Sites with HII < 0.4 are highly impacted (very disturbed), sites with $0.4 \leq$ HII < 0.8 are moderately impacted (moderate disturbance), and those with HII ≥ 0.8 were minimally impacted.

Indicator species: To determine species that are indicators for each of the categories of habitat integrity, based on the collected data, I selected species of high specificity and high fidelity to habitats (Dufrene & Legendre, 1997; Dutra & Marco, 2015; Siddig et al., 2016). Selection was performed using the "indval" function in the "labdsv" package (Dufrêne and Legendre1997) of R (R Core Team 2020). The input variables consisted of the three HII categories and odonate species recorded in each category. Indicator Values are based on the principle described in the following formula:

$$\mathbf{IndVal_{yx} = Specificity_{yx} * Fidelity_{yx} * 100}$$

where $IndVal_{yx}$ is a "y" species in relation to a "x" type of site, $Specificity_{yx}$ is the proportion of sites of type "x" with species "y", and $Fidelity_{yx}$ is the proportion of the number of individuals (abundance) of species "y" that are in a "x" type of site. This allowed me to associate each site with one of the three levels of habitat integrity and calculate an indicator value for each site. The "indval" function produced a list of indicator species arranged in decreasing order of indicator value (indval) (Table 5-4).

To understand the extent to which agriculture and mining might impact freshwater habitats, I carried out a series of tests listed in Table 1-4.

Table 1-4: Tests and analyses conducted to understand the extent to which land uses are associated with freshwater habitat quality

Analysis Variables	**Agricultural and mining sites compared to their reference sites**	**Comparison between agricultural and mining sites**
DBI	t-test to compare DBI site values of agricultural and mining sites to their respective reference sites and plotted a histogram of DBI in each (Figure 3-4).	t-test to compare DBI site values between agricultural sites and mining sites and plotted a histogram of DBI in each (Figure 3-4).
Specific indicator species		t-test to compare abundance of four indicator species between the two land uses (Table 7-4).
Physical-chemical and environmental variables		t-test to compare physical-chemical variables between agricultural and mining sites (Table 8-4)
Nutrients (Nitrite and Phosphates)		t-test and box plots to assess differences in nutrient concentrations (Nitrite and Phosphates) between agriculture and mining sites (Figure 4-4).

Species clustering based on their preferences to environmental variables

This clustering analyzed the most abundant dragonflies separately from damselflies. Dragonfly considered the most abundant have a frequency of at least 15. These are *Brachythemis leucosticta, Nesciothemis farinosa, Orthetrum brachiale* and *Pantala flavescens*, as well as species of damselflies with more than 200 observations, which are *Agriocnemis gratiosa, Ceriagrion glabrum, Proischnura subfurcata, Pseudagrion kersteni* and *Pseudagrion* *spernatum*. In order to visually analyze differences in occurrence of these species I used Principal Component Analysis (PCA) with the prcomp function in R (R Core Team 2020). PCA reduced the dimensionality of predictor variables (listed in Table 2-4) and uncorrelated variables were used to create clusters of similar species in terms of predictor variables. This was plotted using ggplot functions (Wickham, 2016; Abdi, 2010) . I analyzed dragonflies separately from damselflies. The predictor variables included in this analysis are those that have previously been reported to be the most influential to odonate species composition in a habitat (Maltchik et al., 2010; Davis et al., 2018).

Table 2-4: Variables included in the Principle Component Analysis. These are variables sampled in each site

Categories	Variables
Physical-chemicals	pH
	Dissolve Oxygen, DO (%)
	Conductivity (µS/cm)
	Water Temperature (^{0}C)
	Turbidity (cm)
	Longitude
Geographic coordinates	Latitude
	Elevation
Substrates in water body	Mud (%)
	Silt (%)
	Gravel (%)
	Rocks (%)
	Sand (%)
	Detritus (%)
	Deadwood (%)
	Macrophytes in water (%)
Riparian condition	Riparian canopy cover (%)
Air conditions	Air humidity
	Air temperature

	Wind speed

Influential bioclimatic and hydrological variables and prediction of the most abundant odonates

To predict the occurrence of the most abundant odonate species (damselflies that had over 200 records and dragonflies that had 150 records), the maximum entropy distribution approach was performed using MaxEnt model (Phillips & Dudik, 2008; Zare et al., 2016). The input variables to this model consisted of bioclimatic and hydrological variables (Table 3-4). These variables were selected based on their biological relevance to odonate distribution (Marques et al., 2018; Waszkowiak et al., 2002). Predictive performance of the model was assessed using the Area Under Curve (AUC) (Marques et al., 2018; Zare et al., 2016). The model agreement of performance ranges from 0 to 1: <0.05; very poor: 0.05−0.20; poor: 0.20−0.40; fair: 0.40−0.55; good: 0.55−0.70; very good: 0.70−0.85; excellent: 0.85−0.99; to perfect: 0.99−1.00.

Bioclimatic variables (Table 3-4) were downloaded from Worldclim (https://www.worldclim.org/data/bioclim.html), and three hydrological variables from HydroSHEDS (https://hydrosheds.cr.usgs.gov/hydro.php) (Hugo et al., 2012; Lehner et al., 2008). A 0.90 km buffer distance from each sample point was added in ArcGIS. To reduce bioclimatic variables to variables that are not highly correlated r<|0.90|, Pearson correlation test was performed to remove the highly correlated variables (Good, 2009; Sallis et al., 1997). For the highly correlated variables r>|0.90|, one representative variable was considered. In the end, the original 23 variables sampled were reduced to 11 variables (Table 3-4).

Using ArcGIS 10.6.1, layers of 11 bioclimatic and hydrologic variables were created at resolution of 1 km^2 (Table 3-4). In addition, to downweigh the densely sampled areas, a Gaussian kernel density sampling bias file of the same resolution was created using a distance buffer of 0.9km2 using SDMtoolbox (Brown, 2014).

Furthermore, through the MaxEnt model, the percentage of contribution of bioclimatic and hydrological variables to occurrence of each species was determined in the model using the jackknife test of MaxEnt (Steven et al., 2008; Waszkowiak et al., 2002). This allowed identification of variables with the most influence on the occurrence of different odonate species.

Table 3-4: Bioclimatic variables extracted from Worldclim (grey rows) and hydrological and other variables from HydroSHEDS (white rows).

	Variables used in MaxEnt Model
1	BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp))
2	BIO3 = Isothermality (BIO2/BIO7) (×100)
3	BIO4 = Temperature Seasonality (standard deviation ×100)
4	BIO12 = Annual Precipitation
5	BIO15 = Precipitation Seasonality (Coefficient of Variation)
6	BIO18 = Precipitation of Warmest Quarter
7	BIO19 = Precipitation of Coldest Quarter
8	Conditioned digital elevation
9	Flow direction (Flwdir)
10	Flow accumulation (Flwacc)
11	Distance to water

Results

Overall, results showed that there was no significant difference between DBI site values of agricultural and mining sites ($t = 0.99$, $df = 21.89$, p-value > 0.05), however, there was a significant difference between agricultural sites and their references ($t = -5.21$, $df = 7.98$, p-value < 0.001) and between mining and reference sites ($t = 3.26$, $df = 6.42$, p-value < 0.05). Riparian vegetation was significantly different between reference sites and those with both land use practices. Additionally, agriculture was significantly associated with higher water conductivity ($t = 2.37$, $df = 11.18$, p-value <0.05), while mining was significantly associated with the increased water turbidity ($t = 2.97$, $df = 7.95$, p-value< 0.05) and more sandy substrates ($t = -2.6026$, $df = 12.13$, p-value< 0.05).

The PCA suggested that the variance explained by the two axes, PC1 and PC2, is higher in the most abundant damselflies than in dragonflies. While the most abundant dragonfly species do not show a clear separation (Figure 5-4), two species of the most abundant damselflies, *Agriocnemis gratiosa* and *Ceriagrion glabrum* exhibit the highest separation from others (Figure 6-4). i.e these two species are more closely related in terms of habitat preference based on their predictor variables (Physical-chemical and environmental variables). As for the influence of bioclimatic and hydrological variables species occurrence, the jackknife procedure showed that the precipitation of the coldest quarter, conditioned elevation, and flow accumulation are the biggest factors to occurrence of the most abundant damselflies and dragonflies (Table 3-4).

Table 4-4: The summary of the analyzed covariates and the results

Attribute	**Agricultural and mining sites compared to their reference sites**	**Comparison between agricultural and mining sites**	**Occurrence prediction of the most abundant species**

DBI	Significant differences Agriculture: t = -5.21, df = 7.98, p-value < 0.01 Mining: t = 3.26, df = 6.42, p-value < 0.05 (Figure 3-4)	Not significant: t = 0.99, df = 21.89, p-value > 0.05 (Figure 3-4)	
Specific Indicator species		Mining sites: Abundance of *Pseudagrion kersteni,* (t = 0.76, df = 9.56, p-value > 0.05) Agricultural sites: Abundance of *Trithemis arteriosa* (t = 2.66, df = 6, p-value < 0.05) (Table 7-4)	
Physical-chemical and Environmental variables	Agriculture: Significant electric conductivity (t = 2.37, df = 11.18, p-value < 0.03). Riparian canopy (%) (t = 2.26, df = 12.49, p-value < 0.05). Mining: Significant difference in turbidity (t = 2.9775, df = 7.95, p-value < 0.05), sand (%) (t = -2.60, df = 12.13, p-value < 0.05) and riparian canopy (%) (t = 2.26, df = 12.49, p-value < 0.05) (Tables 7-4)		The most abundant dragonflies did not show different patterns in environmental preference, while damselflies exhibit different patterns: *Agriocnemis gratiosa* and *Ceriagrion glabrum* are closely related in terms of their predictor variables, dragonflies did not show a clear pattern (Figure 5-4 & 6-4).
Nutrients		Non-significant difference Nitrites: t = 1.44, df = 8, p-value> 0.05 Phosphates: t = -0.33, df = 10, p-value > 0.05 (Figure 4-4)	

Bioclimatic and hydrological variables			The most influential variables to species occurrence are: Precipitation of the coldest quarter, conditioned elevation and flow accumulation (Table 8-4)

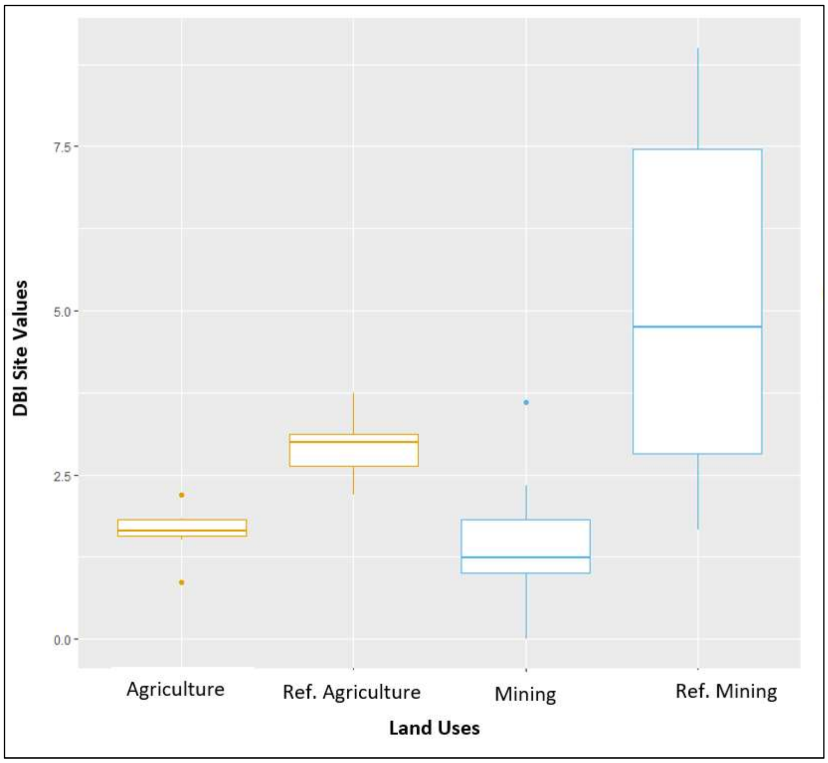

Figure 3-4. DBI site values of study sites categorized by land use type: agricultural sites and agricultural reference sites, mining sites and mining reference sites. There is a significant difference between agricultural and its reference sites ($t = -5.21$, $df = 7.98$, p-value < 0.001), as well as mining and its reference sites ($t = 3.26$, $df = 6.42$, p-value < 0.05). However, the difference between Agricultural and mining sites is not significant ($t = 0.99$, $df = 21.89$, p-value > 0.05).

Table 5-4: Odonate indicator species per habitat category of human impact. For each habitat integrity category, species are listed in a decreasing order of indicator values. All the listed species significantly reflect the category of habitat integrity they are associated with based on p-value of the model computed using Indicator Value (IndVal) function in R.

Species	**Categories of Habitat Integrity**	**Indicator Value**	**p-Value**
Pantala flavescens	High Impact	0.25	< 0.05
Anax imperator	High Impact	0.23	< 0.05
Pseudagrion kersteni	High Impact	0.21	< 0.05
Africallagma elongatum	High Impact	0.14	< 0.05
Trithemis arteriosa	Moderate Impact	0.25	< 0.05
Pseudagrion sublacteum	Moderate Impact	0.18	< 0.05
Pseudagrion massaicum	Moderate Impact	0.15	< 0.05
Trithemis annulata	Moderate Impact	0.15	< 0.05
Zosteraeschna ellioti	Minimal Impact	0.19	< 0.05
Africallagma pseudelongatum	Minimal Impact	0.15	< 0.05
Tramea basilaris	Minimal Impact	0.13	< 0.05
Atoconeura pseudeudoxia	Minimal Impact	0.10	< 0.05

Table 6-4: Species that are only recorded in relatively pristine habitats. These species can be considered as indicators of intact habitat in respect to ecological zones they inhabit.

Species	**Categories of Habitat Integrity**	**Ecological zone**	**Habitat**
Pseudagrion kamiranzovu	Pristine habitats	South West	Nyungwe National Park
Atoconeura pseudeudoxia	Pristine habitats	South West	Nyungwe National Park
Neodythemis Nyungwe	Pristine habitats	South West	Nyungwe National Park
Stenocypha jacksoni	Pristine habitats	South West	Nyungwe National Park
Stenocypha tenuis	Pristine habitats	South West	Nyungwe National Park-Cyamudongo
Atoconeura pseudeudoxia	Pristine habitats	South West	Nyungwe National Park-Cyamudongo
Diplacodes pumila	Pristine habitats	North Central	Rugezi wetland
Notogomphus lujai	Pristine habitats	North West	Gishwati wetlands
Agriocnemis palaeforma	Pristine habitats	North East	Akagera National Park

Tetrathemis camerunensis	Pristine habitats	North East	Akagera National Park

Table 7-4: Comparison of abundance odonate indicator species for high impact and moderate impact abundance mining and agricultural sites. * Indicates species with a significant difference in abundance

Species	P-Value	Mean		SD	
		Mining	Agriculture	Mining	Agriculture
Pantala flavescens	$t = 1.70$, $df = 6$, p-value > 0.05	0.00	4.857	0.00	7.53
Pseudagrion kersteni	$t = 0.76$, $df = 9.56$, p-value > 0.05	5.3	8.714	6.73	10.35
Pseudagrion sublacteum	$t = 1$, $df = 6$, p-value > 0.05	0.00	1.429	0.00	3.77
Trithemis arteriosa*	$t = 2.66$, $df = 6$, p-value < 0.05	0.00	3.429	0.00	3.408

The comparison of physical-chemical and riparian zone variables shows differences between land uses and reference sites (Table 7-4). For both land uses, significant differences are noted in water turbidity, electric conductivity, sandy substrates and canopy cover (Table 4-4). The comparison of nutrients (nitrites and phosphates) in agriculture and mining sites did not show significant differences (Nitrites: $t = 1.44$, $df = 8$, p-value $= 0.18$; Phosphates: $t = -0.33$, $df = 10$, p-value $= 0.74$) (Figure 4-4).

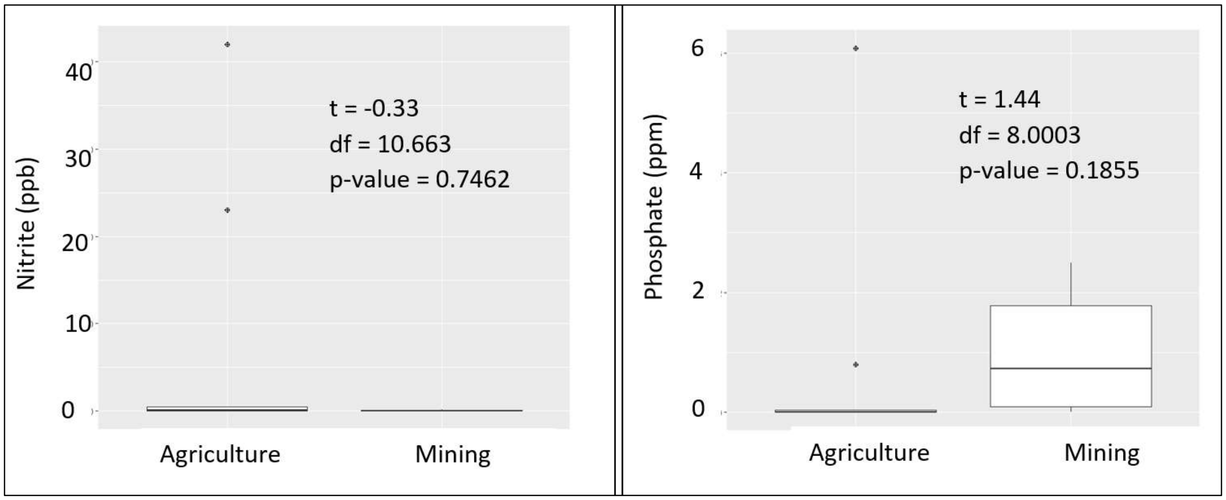

Figure 4-4. Comparison of nutrient concentrations (nitrites and phosphates) in agricultural and mining sites. For nitrites, concentrations in agricultural sites and mining were close to zero. Phosphate concentration was also close to zero. Outliers (dots) of concentrations of nitrites and phosphates were recorded in agricultural sites for both nitrite and phosphate concentrations.

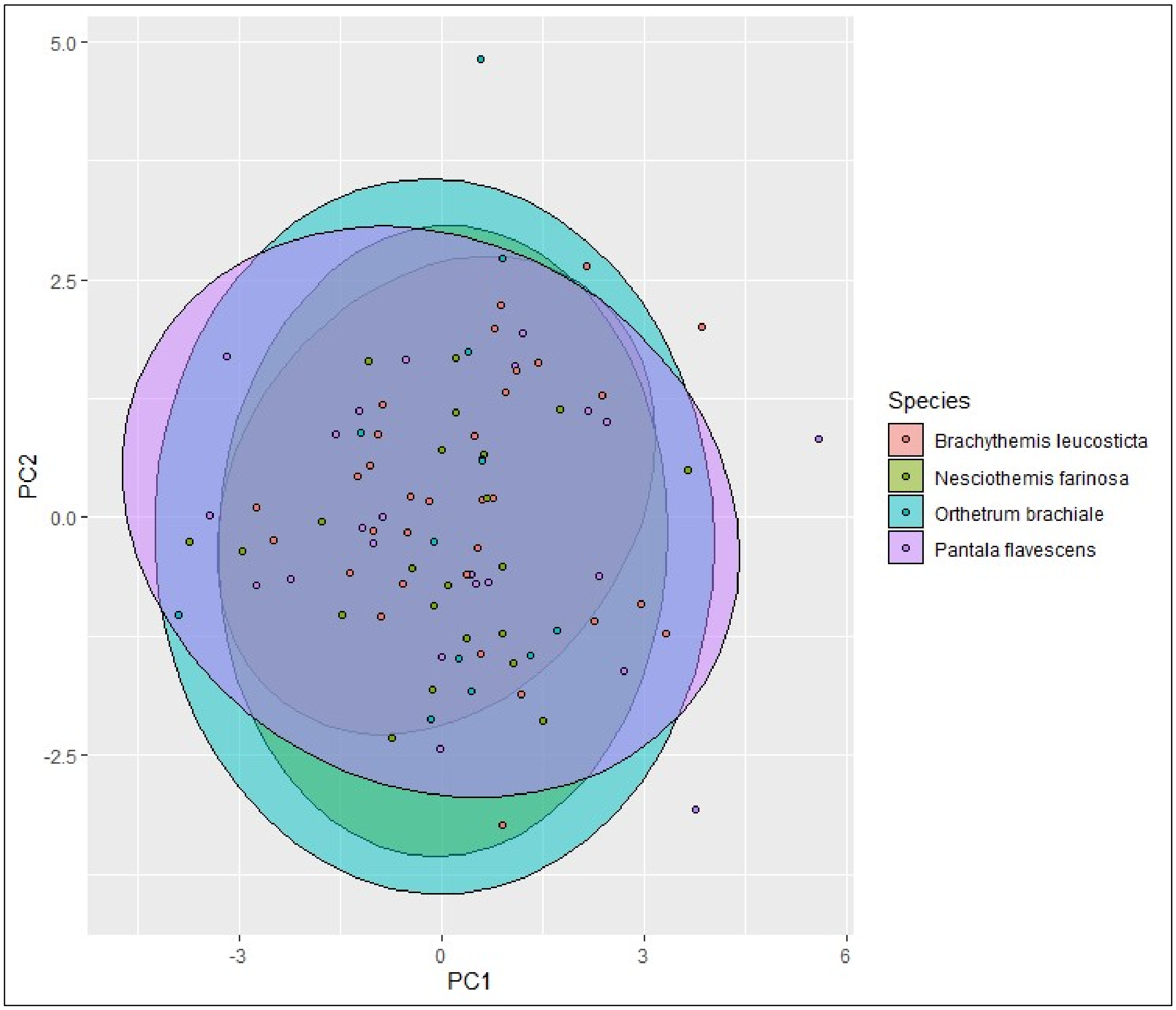

Figure 5-4. Clusters of the most abundant dragonfly species based on their association to predictor variables (physical-chemical and environmental variables reduced to two first principle components). The dots represent each species position as determined the first and second principle component (PC1 &PC2). The ellipses cover each species position in terms of the two principle components.

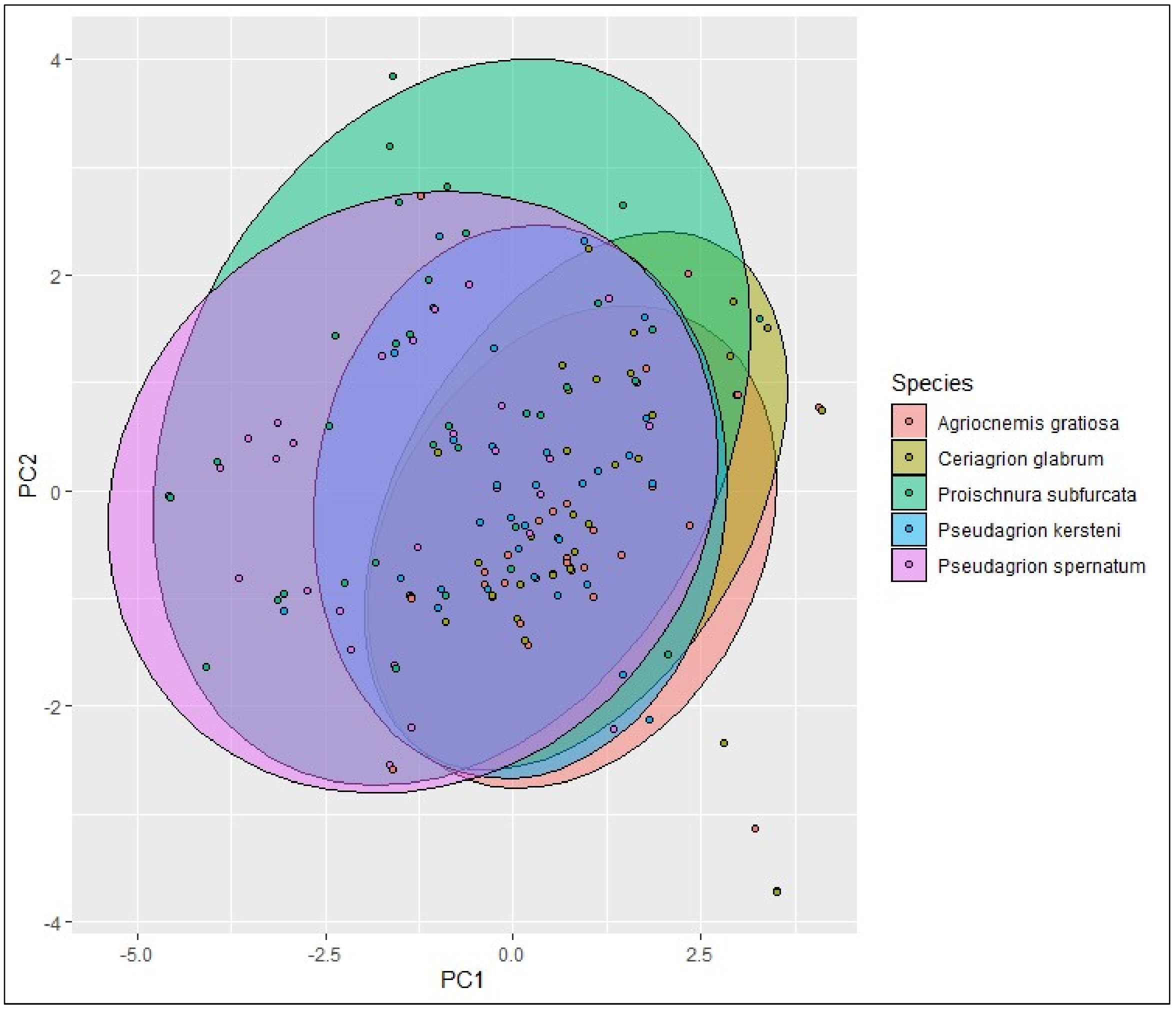

Figure 6-4. Clusters of the most abundant damselfly species based on their association to predictor variables (physical-chemical and environmental variables reduced to PC1 and PC1). The dots represent each species 'position as determined by the first and second principle component (PC1 & PC2). *Agriocnemis gratiosa* and *Ceriagrion glabrum* are closely related in terms of their predictor variables.

Table 8-4: Jackknife procedure in MaxEnt showing the percentage of influence of each covariate to species distribution. The sum of each row should be 100. i.e., each variable contributes differently to occurrence of species. The row (*) for covariates that are > 20, which is considered here as covariate of high influence. Columns with dark cells represent variables that have higher influence to more than two species.

	Bio12	**Bio15**	**Bio18**	**Bio19**	**Bio2**	**Bio3**	**Bio4**	**Codem**	**Flw_dir**	**Flwacc**	**Rw_dist**
Agriocnemis gratiosa	1.10	1.84	4.99	31.65*	3.36	17.25	1.78	22.50*	1.32	7.98	6.22
Brachythemis leucosticta	3.32	3.47	7.17	20.53*	4.22	1.75	0.18	16.18	2.82	33.00*	7.36
Ceriagrion glabrum	2.72	1.22	1.89	21.36*	3.36	0.72	0.59	33.39*	3.38	6.60	24.77*
Nesciothemis farinosa	8.67	0.23	5.63	13.47	0.18	7.59	0.40	0.25	25.12*	35.52*	2.94
Orthetrum brachiale	4.49	0.16	6.51	37.06*	1.65	0.32	0.39	32.66*	2.70	12.01	2.04
Pantala flavescens	7.38	4.55	7.36	8.16	1.23	5.16	3.05	12.32	7.18	41.10*	2.51
Proischnura subfurcata	3.83	0.04	56.81*	4.33	0.24	6.60	2.52	0.66	5.16	16.44	3.37
Pseudagrion kersteni	7.44	0.32	7.77	5.30	0.24	15.34	0.46	24.79*	11.83	19.93	6.56
Pseudagrion spernatum	22.60*	6.79	5.58	9.06	2.94	5.06	1.00	0.30	11.01	33.40*	2.26

Bio12 = Annual Precipitation; Bio15 = Precipitation Seasonality; (Coefficient of Variation); Bio18 = Precipitation of Warmest Quarter; Bio19 = Precipitation of Coldest Quarter; Bio2 = Mean Diurnal Range (Mean of monthly (max temp - min temp)); Bio3 = Isothermality; codem =Conditioned elevation, flwacc= flow accumulation; flw_dir=flow direction; rw_dist=distance to water

Discussion

The present study highlights the shift in ecological conditions associated with agriculture and mining based on dragonfly biotic index, individual odonate species, physical-chemical variables and physical chemical variables. This is consistent with earlier studies that suggested that land conversion often leads to changes in biotic structure and composition in one way or another, affecting habitat integrity (Kietzka et al., 2018; Walsh et al, 2007). This shows effectiveness of using odonate based-indices to assess the relationship of agriculture and mining activities freshwater habitat integrity. Here, I discuss the potential for using odonates as indicators in immediate, medium and long-term monitoring.

Using Dragonfly Biotic Index in habitat assessment

Dragonfly Biotic Index has previously shown potential to effectively assess habitat quality. It provides mean to monitor for threats such as habitat degradation, pollution species invasion and climate (McGeoch et al., 2011) Simaika & Samways, 2009). My findings suggest that both mining and agriculture are associated to degradation of freshwater ecosystems based on results of the Dragonfly Biotic Index site values. These values are significantly higher in reference sites than agricultural or mining sites. DBI can be handy in evaluating the ecological changes over time in relation to changes in land use practices. In the same context, DBI could be used to compare these land use practices with benchmark sites (relatively pristine sites). This could provide an information about the extent to which land use practices differ from each other, or change over time in terms of ecological integrity.

Using odonates in immediate impact assessment

To get a better sense of the magnitude of the impacts of agriculture and mining, this study used specific indicator species. Previous studies have suggested that assessing species of high fidelity, i.e., abundant species within a habitat of specific habitat integrity level, is useful in long-term monitoring of habitats (Ball-Damerow et al., 2014). While the results of the present study suggest that there is no significant difference between agriculture and mining based on DBI, individual indicator species show a significant higher abundance of species that indicate moderate impact in agricultural sites than in mining sites (*Trithemis arteriosa*) and the absence of other indicator species. In order words, this could mean that Agriculture presents a moderate impact while mining shows high impact to freshwater habitats. The absent of other indicator species and the slightly higher abundane *Pseudagrion kersteni*, the indicator of "high impact" sites, suggests that *highly impacted* category of freshwater habitats is associated with mining. In agricultural sites, the absence of other indicator species and significant higher abundance of an indicator of moderate habitat integrity (*Trithemis arteriosa*). Individual indicator species could therefore be useful in in teasing out different levels of habitat integrity in relationship to various land use types.

In this study, indicator species enabled analysis to move a step further to estimate the trend of covariates and specification of the degree of human impact (high impact and moderate impact). Similar to this study suggesting that *Trithemis arteriosa and Pseudagrion sublacteum* indicate habitats with moderate impacts, *Trithemis arteriosa* was previously shown to be an indicator for permanent water bodies like reedy pools, streams or swamps that are in fairly good ecological conditions (Giere & Hadrys, 2006), and earlier work found *Pseudagrion* *sublacteum* to be more associated to habitats with moderately disturbances than those that are intact (Cotgreave& Forseth, 2009).

This study suggested that *Pantala flavescens Pseudagrion kersteni*, *Pseudagrion spernatum* are among indicator species for highly human impact. This is consistent with previous studies. *As its* vernacular name suggests, “wandering glider”, *Pantala flavescens* migration is the furthest known migration of any known insect. Also, earlier studies on large scale of biotope gradients suggested *Pantala flavescens* is among the eurytopic odonate species i.e generalist. (Devaud & Lebouvier, 2019;M.J.Samways, 1996). *Pseudagrion kersteni*, *Pseudagrion spernatum* have been found to be abundant in open habitats, which is often subsequent to habitat degradation such as riparian removal (Dijkstra et al., 2007; Pereira et al., 2019).

Additionally, previous studies suggested that specific odonate indicator species could be a good way to translate habitat integrity into magnitudes of impacts and reflect impacts in medium-term (Miguel et al., 2017). Also, specific indicator species should be based on two criteria. First, species must display high specificity, whereby they are abundant within a specific type of habitat. Second, they must display good site fidelity, meaning that they consistently occur in that same habitat over time and space (Miguel et al., 2017; Rocha-ortega et al., 2019). Therefore, the selected indicator species in this study were abundant and consistently found within a habitat with specific ecological conditions or level of disturbance.

The use of specific indicator species may replace the need to sample entire communities, hence, less time consuming (Miguel et al., 2017; Monteiro, Juen, & Hamada, 2015). The use of DBI in conjunction with specific indicator species can, therefore, provide both a more robust and accurate technique, and a relatively easily interpretable indication of ecological conditions (Dutra & Marco, 2015; Siddig et al., 2016).

Changes in assemblages of odonates can reflect variations in environmental and physical- chemical variables, most of which are immediate responses to threats (Dodds et al., 2013; Va & Favila, 2017). Previous studies have suggested that odonates can be used at the sub-order and genus taxonomic level to indicate habitat integrity. Zygoptera (damselflies) were found to be sensitive to disturbance and tended to have higher abundance in less disturbed habitats when compared to anisoptera (dragonflies), which are generally tolerant to habitat disturbance and showed higher abundance in disturbed habitats (Marques et al., 2018; Miguel et al., 2017). However, these patterns were not observed in the present study. There was no difference in abundance of anisoptera and zygoptera, when comparing areas of high, moderate and minimal human impacts.

Odonates as indicators of changes in physical-chemical variables in relation to agriculture and mining in wetlands

This study has found differences in physical and chemical factors associated with mining and agriculture as reflected in the change of the structure of odonate assemblages and DBI. Mining contributed to accumulation of sandy substrates and increased water turbidity. My findings align with previous studies that suggest that open-pit mining causes changes in habitat structure and alters the integrity of aquatic habitats as it consists of digging out sand and encroaching on water body bed, which increases sediment loads that damage aquatic life (Dedieu et al., 2015). These practices cause shifts in habitat structure and disturb breeding mechanisms and shelter sites for aquatic biota. Additionally, high turbidity suffocates odonates by covering their gills (Dedieu et al., 2015; Sievers et al., 2018), which consequently affects the physiological performance and reproductive success (García-García et al., 2017).

The results of this study show significant changes in riparian vegetation caused by agriculture and mining. Agricultural and mining activities result in decreases in canopy cover which is often related to the removal of riparian vegetation (Dodds et al., 2019), leading to deficiencies in the riparian role. The benefits of riparian vegetation include the stabilization of water body banks, protection of the soil surface from erosion, prevention of the water body from heating, and filtration of upslope run offs (Dosskey et al., 2010; Venson et al., 2017). Loss of riparian vegetation causes weakening bank stability and increasing risk of erosion and flooding, which amplify siltation and sediments (Cunha et al., 2019; Royer et al., 2008). Additionally, the slight increase of water temperature caused by agriculture may be explained by the removal of riparian vegetation as suggested by previous studies (García-García et al., 2017; Salmah et al., 2006).

The degradation of riparian zones, as documented in this study, may be a key element in exacerbating the pollution of freshwater habitats. Agriculture is known to contribute to increases in nutrient concentrations in water resulting from the use of fertilizers and degradation of riparian zones which help filter run off (Dosskey et al., 2010; Cunha et al., 2019). Most likely due to the fact that all mining sites were not completely free from agricultural effects, there was no significant difference in nutrient concentrations between agricultural and mining sites. However, the close proximity of agricultural study sites may have caused the recorded outliers, which show extremely high concentrations of phosphates and nitrites. This enrichment of nutrients in some agricultural sites may be due to excessive application of fertilizer around those particular sites (Wurtsbaugh et al., 2019). The enrichment of nutrients leads to high concentration ions, which could explain the significant increased electric conductivity recoded in agricultural sites (Mugni, H., Paracampo, A., & Bonetto, 2013). There are multiple consequences of excess nutrients that are worth noting, including algal blooms that can limit sun light penetration to deeper layers of water bodies, which cause depletion of oxygen concentration (García-García et al., 2017; Salmah et al., 2006). The trend of such cascading series of phenomena resulting from eutrophication and degradation, which affect habitats integrity of freshwater habitats, can therefore be pinpointed using odonate-based indices.

Using odonates in long-term monitoring

As previously mentioned, abundant species are good indicators when they have fidelity to a limited range of habitat types; however, widespread species can also be useful in long-term monitoring. As predators, these species are an important component of the trophic web in freshwater ecosystems (Caesar, 2012). Negative changes in odonate communities can therefore indicate modified ecological conditions caused by habitat degradation (eg: agriculture and mining) or climate changes (Berquier et al., 2016;Maltchik et al., 2010). Additionally, once restoration of the degraded habitats is undertaken, evolution of odonate communities can be associated with improvement of ecological conditions (Koch et al., 2014; Modiba et al., 2017).

Individual indicator species of odonates that are easily identifiable with predictable responses to habitat integrity can be used via citizen science as tools to assess habitat recovery over time (Modiba et al., 2017; Ožana et al., 2019). This study provides a list of species that could potentially indicate various stages of ecosystem restoration (Table 5-4, 6-4). For example, species that indicate moderate impact and very distinctive such as *Trithemis arteriosa, Pseudagrion sublacteum, Pseudagrion massaicum and Trithemis annulata* could appear in medium term. Species that reflect minimum impact (Table 5-4) or intact habitat (Table 6-4) could serve as a long-term recovery target. For example, restoration of papyrus wetlands in the peripheries of Kigali city could envisage recovering species *Agriocnemis palaeforma* in a long-term, while wetlands inside Kigali city, species of moderate impact (listed above) could be the target of ecosystem recovery.

Furthermore, my findings show several bioclimatic and hydrological variables that influence the occurrence of widespread, abundant species, such as precipitation of the coldest quarter, conditioned elevation, and flow accumulation. Most of these variables are affected by climate change. Therefore, not only has climate change been found to negatively affect occurrence, but also the phenology and flight performance of odonates (Marques et al., 2018; McCauley et al., 2018). As such, responses of freshwater ecosystems to climate change can be assessed by long-term monitoring of odonate species that are both widespread and abundant. Given the weight of these species in the DBI, any significant changes in their presence may indicate that the DBI scoring system needs to be calibrated.

Conclusion and recommendations

- This study highlights that agriculture and mining negatively affect freshwater ecosystems based on results of odonate-based indices, individual odonate species and physical-chemical variables.
- While mining has apparently higher impacts in wetlands than agriculture, both agriculture and mining contribute to degradation of riparian zones.

- Ecologically friendly practices and restoration of degraded habitats is highly recommended, especially to maintain riparian zones.
- Odonate-based indices could be used to monitor restoration practices and to steer sustainable and environmentally friendly agricultural and mining practices. Odonates also offer ways to monitor ecosystems at different time scales (immediate, medium and long-term).
- The most influential bioclimatic and hydrological variables affecting odonate occurrence are precipitation of the coldest quarter, conditioned elevation, and flow accumulation and these are influenced by variability and climate change.
- Responses of ecosystems to effects of climate change can be monitored by assessing the most common odonate species in a long-term.
- The use of odonate in monitoring can potentially steer sustainable and environmentally friendlier agriculture and mining while preserving the integrity of freshwater ecosystems.

Appendix 1-4: Permission to use data from USGS (HydroSHEDS)

--

USGS EROS User Services

to me

Apr 27, 2020, 11:06 AM (23 hours ago)

Hello Erasme,

Thank you for contacting the USGS Earth Resources Observation and Science (EROS) Center.

Most of the images, data, and related products available from the USGS Earth Resources Observation and Science (EROS) Center, which also contains the NASA Land Processed Distributed Active Archive Center (LP DAAC), are federally created data and therefore reside in the public domain and may be used, transferred, or reproduced without copyright restriction.
https://www.usgs.gov/centers/eros/data-citation?qt-science_support_page_related_con=0#qt-science_support_page_related_con

Best regards,

References

Abdi, H., W. L. J. (2010). Principal component analysis. Wiley interdisciplinary reviews: *Computational Statistics*, *2*(4), 433–459. https://doi.org/10.1002/wics.101

Ball-Damerow, J. E., M'Gonigle, L. K., & Resh, V. H. (2014). Local and regional factors influencing assemblages of dragonflies and damselflies (Odonata) in California and Nevada. *Journal of Insect Conservation*, *18*(6). https://doi.org/10.1007/s10841-014-9709-6

Berquier, C., Orsini, A., Ferrat, L., & Andrei-ruiz, M. (2016). " Odonata Community Index – Corsica " (OCIC): A new biological index based on adult odonate populations for assessment of the ecological status of watercourses in Corsica. *Ecological Indicators*, *66*, 163–172. https://doi.org/10.1016/j.ecolind.2016.01.022

Brown, J. L. (2014). SDMtoolbox: A python-based GIS toolkit for landscape genetic, biogeographic and species distribution model analyses. *Methods in Ecology and Evolution*, *5*(7), 694–700. https://doi.org/10.1111/2041-210X.12200

Butchart, S. H. M., Walpole, M., Collen, B., Strien, A. Van, Jörn, P. W., Almond, R. E. A., … Lamarque, J. (2018). Global biodiversity: indicators of recent declines. *Science*, *328*(5982), 1164-1168. https://doi.org/10.1126/science.1187512

Caesar, R. M. (2012). *Phylogeny of the Genus Argia (Odonata: Coenagrionidae) with Emphasis on Evolution of Reproductive Morphology* (Dissertati). The Ohio State University.

Caniani, D., Labella, A., Lioi, D. S., Mancini, I. M., & Masi, S. (2016). Habitat ecological integrity and environmental impact assessment of anthropic activities: A GIS-based fuzzy logic model for sites of high biodiversity conservation interest. *Ecological Indicators*, *67*, 238–249. https://doi.org/10.1016/j.ecolind.2016.02.038

Clausnitzer, V., Kalkman, V. J., Ram, M., Collen, B., Baillie, J. E. M., Bedjanič, M., … Wilson, K. (2009). Odonata enter the biodiversity crisis debate: The first global assessment of an insect group. *Biological Conservation*, *142*(8), 1864–1869. https://doi.org/10.1016/j.biocon.2009.03.028

Cotgreave, P., & Forseth, I. (2009). *Introductory ecology*. John Wiley & Sons.

Cunha, D. G. F., Finkler, N. R., Gómez, N., Cochero, J., Donadelli, J. L., Saltarelli, W. A., … Thomas, S. A. (2019). Agriculture influences ammonium and soluble reactive phosphorus retention in South American headwater streams. *Ecohydrology*, (November), 1–14. https://doi.org/10.1002/eco.2184

Davis, S. J., Daire, Ó., Mellander, P., Kelly, A., Matthaei, C. D., Piggott, J. J., & Kelly-quinn, M. (2018). Science of the Total Environment Multiple-stressor effects of sediment , phosphorus and nitrogen on stream macroinvertebrate communities. *Science of the Total Environment*, *637–638*, 577–587. https://doi.org/10.1016/j.scitotenv.2018.05.052

Dedieu, N., Rhone, M., Vigouroux, R., & Céréghino, R. (2015). Assessing the impact of gold mining in headwater streams of Eastern Amazonia using Ephemeroptera assemblages and biological traits. *Ecological Indicators*, *52*, 332–340. https://doi.org/10.1016/j.ecolind.2014.12.012

Devaud, M., & Lebouvier, M. (2019). First record of Pantala flavescens (Anisoptera : Libellulidae) from the remote Amsterdam Island , southern Indian Ocean. *Polar Biology*, *42*(5), 1041–1046. https://doi.org/10.1007/s00300-019-02479-3

Dijkstra &, & Clausnitzer. (2014). *The Dragonflies and Damselflies of Eastern Africa: Handbook for all Odonata from Sudan to Zimbabwe*. (298, Ed.). Studies in Afrotropical Zoology. Retrieved from https://www.africamuseum.be/

Dijkstra, K., Groeneveld, L. F., Genetic, N., & Clausnitzer, V. (2007). The Pseudagrion split : Molecular phylogeny confirms the morphological and ecological dichotomy of Africa ' s most diverse genus of Odonata (Coenagrionidae) The Pseudagrion split : molecular phylogeny confirms the morphological and ecological dichotomy , (April). https://doi.org/10.1080/13887890.2007.9748286

Dodds, W. K., Bruckerhoff, L., Batzer, D., Schechner, A., Pennock, C., Renner, E., … Grieger, S. (2019). The freshwater biome gradient framework: Predicting macroscale properties based on latitude, altitude, and precipitation. *Ecosphere*, *10*(7). https://doi.org/10.1002/ecs2.2786

Dodds, W. K., Perkin, J. S., & Gerken, J. E. (2013). Human impact on freshwater ecosystem services: A global perspective. *Environmental Science and Technology*, *47*(16), 9061–9068. https://doi.org/10.1021/es4021052

Dosskey, M. G., Vidon, P., Gurwick, N. P., Allan, C. J., Duval, T. P., & Lowrance, R. (2010). The role of riparian vegetation in protecting and improving chemical water quality in streams. *Journal of the American Water Resources Association*, *46*(2), 261–277. https://doi.org/10.1111/j.1752-1688.2010.00419.x

Dufrene, M. & Legendre, P. (1997). Species assemblages and indicator species : the need for a flexible asymmetrical approach. *Ecological Monographs*, *67*(3), 345–366.

https://doi.org/10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2

Dusková, M., & Machácek, J. (2013). The geomorphological changes caused by the artisanal mining in the mining area Kabera – Rwanda. *International Multidisciplinary Scientific GeoConference: SGEM: Surveying Geology & Mining Ecology Management*, *3*(673.), 673–680.

Dutra, S., & Marco, P. De. (2015). Bionomic differences in odonates and their influence on the efficiency of indicator species of environmental quality. *Ecological Indicators*, *49*, 132–142. https://doi.org/10.1016/j.ecolind.2014.09.016

Fick, S. E., & Hijmans, R. J. (2017). WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. *International Journal of Climatology*, *37*(12), 4302–4315. https://doi.org/10.1002/joc.5086

García-García, P. L., Vázquez, G., Novelo-Gutiérrez, R., & Favila, M. E. (2017). Effects of land use on larval Odonata assemblages in cloud forest streams in central Veracruz, Mexico. *Hydrobiologia*, *785*(1). https://doi.org/10.1007/s10750-016-2900-x

Gerson Araujo, F., Fichberg, I., Teixeira Pinto, B. C., & Galvao Peixoto, M. (2003). A Preliminary Index of Biotic Integrity for Monitoring the Condition of the Rio Paraiba do Sul, Southeast Brazil. *Environmental Management*. https://doi.org/10.1007/s00267-003-3003-9

Giere, S., & Hadrys, H. (2006). Polymorphic microsatellite loci to study population dynamics in a dragonfly, the libellulid Trithemis arteriosa (Burmeister, 1839). *Molecular Ecology Notes*, *6*(3), 933–935. https://doi.org/10.1111/j.1471-8286.2006.01405.x

Golfieri, B., Hardersen, S., Maiolini, B., & Surian, N. (2016). Odonates as indicators of the ecological integrity of the river corridor: Development and application of the Odonate River Index (ORI) in northern Italy. *Ecological Indicators*, *61*, 234–247. https://doi.org/10.1016/j.ecolind.2015.09.022

Good, P. (2009). Robustness of Pearson correlation. *Interstat*, *15*(5), 1–6.

Hilson, G. (2002). Small-scale mining and its socio-economic impact in developing countries. *Natural Resources Forum*, *26*(1), 3–13. https://doi.org/10.1111/1477-8947.00002

Hugo A. S. Guedes; Demetrius D. da Silva. (2012). Comparison between hydrographically conditioned digital elevation models in the morphometric charaterization of watersheds. *Eng. Agríc*, *32*(5). https://doi.org/10.1590/S0100-69162012000500012

Imasiku, K., & Ntagwirumugara, E. (2020). An impact analysis of population growth on energy-water-food-land nexus for ecological sustainable development in Rwanda. *Food and Energy Security*, *9*(1), e185.

Jønsson, J. B., & Bryceson, D. F. (2009). Rushing for Gold : Mobility and Small-Scale Mining in East Africa. *Development and Change*, *40*(2), 249–279.

Jorge L. Nessimian, Eduardo M. Venticinque, Jansen Zuanon, Paulo De Marco Jr, M. G., & Juen, Luana Fidelis, Joana D'arc Batista, L. J. (2011). Odonate biodiversity in terra-firme forest streamlets in Central Amazonia: On the relative effects of neutral and niche drivers at small geographical extents. *Insect Conservation and Diversity*, *4*(4), 265–274. https://doi.org/10.1111/j.1752-4598.2010.00130.x

Khan, A. S., Yi, H., Zhang, L., Yu, X., Mbanzamihigo, E., & Umuhumuza, G. (2019). An integrated social-ecological assessment of ecosystem service benefits in the Kagera River Basin in Eastern Africa. *Regional Environmental Change*, 39–53.

Khatibi, M., & Sheikholeslami, R. (2016). Ecological Niche Theory: A Brief Review. *The International Journal of Indian Psychology*, *3*(2), 42–45.

Kietzka, G. J., Pryke, J. S., & Samways, M. J. (2018). Comparative effects of urban and agricultural land transformation on Odonata assemblages in a biodiversity hotspot. *Basic and Applied Ecology*, (2017). https://doi.org/10.1016/j.baae.2018.08.008

Koch, K., Wagner, C., & Sahlén, G. (2014). Farmland versus forest: Comparing changes in Odonata species composition in western and eastern Sweden. *Insect Conservation and Diversity*, *7*(1). https://doi.org/10.1111/icad.12034

Lehner, B., Verdin, K., Jarvis, A. (2008). New global hydrography derived from spaceborne elevation data. *Eos, Transactions, AGU*, *89*(10), 93-94.

Luke, S. H., Dow, R. A., Butler, S., Khen, C. V. U. N., Aldridge, D. C., Foster, W. A., … Stretton, A. (2017). The impacts of habitat disturbance on adult and larval dragonflies (Odonata) in rainforest streams in Sabah , Malaysian Borneo. *Freshwater Biology*, 491–506. https://doi.org/10.1111/fwb.12880

Maconachie, R., & Conteh, F. M. (2019). Artisanal mining and the rationalisation of informality: critical reflections from Liberia. *Canadian Journal of Development Studies*, (8), 1-18.

Maltchik, L., Stenert, C., Kotzian, C. B., Pires, M. M., Maltchik, L., Stenert, C., & Kotzian, C. B. (2010). Responses of Odonate Communities to Environmental Factors in Southern

Brazil Wetlands. *Journal of the Kansas Entomological Society*, *83*(3), 208–220. https://doi.org/10.2317/JKES0910.13.1

Mangadze, T., Dalu, T., & William Froneman, P. (2019). Biological monitoring in southern Africa: A review of the current status, challenges and future prospects. *Science of the Total Environment*, *648*, 1492–1499. https://doi.org/10.1016/j.scitotenv.2018.08.252

Marques, M., Eduardo, P., Samuel, P., & Göran, R. (2018). Predicting the effects of future climate change on the distribution of an endemic damselfly (Odonata , Coenagrionidae) in subtropical South American grasslands. *Journal of Insect Conservation*, 303–319. https://doi.org/10.1007/s10841-018-0063-y

McCauley, S. J., Hammond, J. I., & Mabry, K. E. (2018). Simulated climate change increases larval mortality, alters phenology, and affects flight morphology of a dragonfly: *Ecosphere*, *9*(3). https://doi.org/10.1002/ecs2.2151

McGeoch, M. A., Sithole, H., Samways, M. J., Simaika, J. P., Pryke, J. S., Picker, M., … Hamer, M. (2011). Conservation and monitoring of invertebrates in terrestrial protected areas. *Koedoe*, *53*(2), 1–13. https://doi.org/10.4102/koedoe.v53i2.1000

Miguel, T. B., Oliveira-Junior, J. M. B., Ligeiro, R., & Juen, L. (2017). Odonata (Insecta) as a tool for the biomonitoring of environmental quality. *Ecological Indicators*, *81*(June), 555–566. https://doi.org/10.1016/j.ecolind.2017.06.010

Modiba, R. V., Joseph, G. S., Seymour, C. L., Fouché, P., & Foord, S. H. (2017). Restoration of riparian systems through clearing of invasive plant species improves functional diversity of Odonate assemblages. *Biological Conservation*, *214*. https://doi.org/10.1016/j.biocon.2017.07.031

Monteiro-Júnior, C. S., Juen, L., & Hamada, N. (2014). Effects of urbanization on stream habitats and associated adult dragonfly and damselfly communities in central Brazilian Amazonia. *Landscape and Urban Planning*, *127*(1), 28–40. https://doi.org/10.1016/j.landurbplan.2014.03.006

Monteiro, S., Juen, L., & Hamada, N. (2015). Analysis of urban impacts on aquatic habitats in the central Amazon basin : Adult odonates as bioindicators of environmental quality. *Ecological Indicators*, *48*, 303–311. https://doi.org/10.1016/j.ecolind.2014.08.021

Mugni, H., Paracampo, A., & Bonetto, C. (2013). Nutrient concentrations in a pampasic first order stream with different land uses in the surrounding plots (Buenos Aires, Argentina). *Bulletin of Environmental Contamination and Toxicology*, *91*(4), 391–395. https://doi.org/10.1007/s00128-013-1079-3

Nsengimana, V. (2017). Perceptions of Local People on the Use of Nyabarongo River Wetland and Its Conservation in Rwanda Perceptions of Local People on the Use of Nyabarongo River. *Society & Natural Resources*, *30*(1), 3–15. https://doi.org/10.1080/08941920.2016.1209605

Ožana, S., Burda, M., Hykel, M., Malina, M., Prášek, M., Bárta, D., & Dolný, A. (2019). Dragonfly Hunter CZ: Mobile application for biological species recognition in citizen science. *PLoS ONE*, *14*(1). https://doi.org/10.1371/journal.pone.0210370

Patz, J. A., Daszak, P., Tabor, G. M., Aguirre, A. A., Pearl, M., Epstein, J., … Zakarov, V. (2004). Unhealthy landscapes: Policy recommendations on land use change and infectious disease emergence. *Environmental Health Perspectives*, *112*(10), 1092–1098. https://doi.org/10.1289/ehp.6877

Pereira, D. F. G., de Oliveira Junior, J. M. B., & Juen, L. (2019). Environmental changes promote larger species of Odonata (Insecta) in Amazonian streams. *Ecological Indicators*, *98*(October 2018), 179–192. https://doi.org/10.1016/j.ecolind.2018.09.020

Peyre, M. K. La, Reams, M. A., & Mendelssohn, I. A. (2001). Linking actions to outcomes in wetland management : An overview of U.S. State wetland management. *Wetlands*, *21*(1), 66–74.

Phillips, S. J., & Dudik, M. (2008). Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. *Ecography*, *31*(2), 161–175.

Remsburg, A. J., & Turner, M. G. (2009). Aquatic and terrestrial drivers of dragonfly (Odonata) assemblages within and among north-temperate lakes. *Journal of the North American Benthological Society*, *28*(1), 44–56. https://doi.org/10.1899/08-004.1

Rocha-ortega, M., Rodríguez, P., & Córdoba-aguilar, A. (2019). Can dragonfly and damselfly communities be used as bioindicators of land use intensification? *Ecological Indicators*, *107*(February). https://doi.org/10.1016/j.ecolind.2019.105553

Rothenberg, S. E., Windham-Myers, L., & Creswell, J. E. (2014). Rice methylmercury exposure and mitigation: A comprehensive review. *Environmental Research*, *133*, 407–423. https://doi.org/10.1016/j.envres.2014.03.001

Royer, T. V., David, M. B., Gentry, L. E., Mitchell, C. A., Starks, K. M., Heatherly, T., & Whiles, M. R. (2008). Assessment of Chlorophyll- a as a Criterion for Establishing Nutrient Standards in the Streams and Rivers of Illinois . *Journal of Environmental Quality*, *37*(2), 437–447. https://doi.org/10.2134/jeq2007.0344

Sallis, J. F., Johnson, M. F., Calfas, K. J., Caparosa, S., & Nichols, J. F. (1997). Assessing perceived physical environmental variables that may influence physical activity. *Research Quarterly for Exercise and Sport*, *68*(4), 345–351.

Salmah, M. R. C., Tribuana, S. W., & Hassan, A. A. (2006). The population of Odonata (dragonflies) in small tropical rivers with reference to asynchronous growth patterns. *Aquatic Insects*, *28*(3), 195–209. https://doi.org/10.1080/01650420600922315

Samways, M. J., Caldwell, P. M., & Osborn, R. (1996). Spatial patterns of dragonflies (Odonata) as indicators for design of a conservation pond. *Odonatologica*, *25*(2). Retrieved from http://natuurtijdschriften.nl/record/592157

Samways, M, J., & Simaika, J, P. . (2014). Manual of Freshwater Assessment for South Africa: Dradonfly Biotic Index. *Suricata 2*, *1*. https://doi.org/10.1017/CBO9781107415324.004

Siddig, A. A. H., Ellison, A. M., Ochs, A., Villar-Leeman, C., & Lau, M. K. (2016). How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. *Ecological Indicators*. https://doi.org/10.1016/j.ecolind.2015.06.036

Sievers, M., Hale, R., Parris, K. M., & Swearer, S. E. (2018). Impacts of human-induced environmental change in wetlands on aquatic animals. *Biological Reviews*, *93*, 529–554. https://doi.org/10.1111/brv.12358

Simaika, J. P., & Samways, M. J. (2009). An easy-to-use index of ecological integrity for prioritizing freshwater sites and for assessing habitat quality. *Biodiversity and Conservation*, *18*(5), 1171–1185. https://doi.org/10.1007/s10531-008-9484-3

Somma, P. (2015). Rwanda's urbanisation policy: A critical reading. *Open House International*, *40*(4), 5–9.

Stewart, D. A. B., & Samways, M. J. (1998). Conserving Dragonfly (Odonata) Assemblages Relative to River Dynamics in an African Savanna Game Reserve, *12*(3), 683–692.

Tichanek, F., & Tropek, R. (2016). The endangered damselfly Coenagrion ornatum in post-mining streams : population size , habitat requirements and restoration, 701–710. https://doi.org/10.1007/s10841-016-9902-x

Uwimana, A., van Dam, A. A., Gettel, G. M., & Irvine, K. (2018). Effects of agricultural land use on sediment and nutrient retention in valley-bottom wetlands of Migina catchment, southern Rwanda. *Journal of Environmental Management*, *219*, 103–114. https://doi.org/10.1016/j.jenvman.2018.04.094

Uyizeye, E., Kaplin A.B., Willey, L., Clausnitzer, V., Kipping J.,& Dijkstra, K. D. B. (n.d.). Developing an odonate-based tool for monitoring freshwater ecosystems in Rwanda. *Chapter 3 of Doctoral Dissertation (Working Paper in Press)*, *Antioch Un*.

Va, G., & Favila, M. E. (2017). Effects of land use on larval Odonata assemblages in cloud forest streams in central Veracruz , Mexico. *Hydrobiologia*, 19–33. https://doi.org/10.1007/s10750-016-2900-x

Vanacker, M., Wezel, A., Oertli, B., & Robin, J. (2018). Water quality parameters and tipping points of dragonfly diversity and abundance in fishponds. *Limnology*, (19), 321–333. https://doi.org/10.1007/s10201-018-0549-z

Venson, G. R., Marenzi, R. C., & Almeida, T. C. M. (2017). Restoration of areas degraded by alluvial sand mining : use of soil microbiological activity and plant biomass growth to assess evolution of restored riparian vegetation. https://doi.org/10.1007/s10661-017-5852-3

Vorster, C., Samways, M. J., Simaika, J. P., Kipping, J., Clausnitzer, V., Suhling, F., & Dijkstra, K. B. (2020). Development of a new continental-scale index for freshwater assessment based on dragonfly assemblages. *Ecological Indicators*, *109*(April 2019), 105819. https://doi.org/10.1016/j.ecolind.2019.105819

Voß, K., & Schäfer, R. B. (2017). Taxonomic and functional diversity of stream invertebrates along an environmental stress gradient. *Ecological Indicators*, *81*(May), 235–242. https://doi.org/10.1016/j.ecolind.2017.05.072

Walsh, C. J., Waller, K. A., Gehling, J., & Mac Nally, R. (2007). Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. *Freshwater Biology*, *52*(3), 574–587. https://doi.org/10.1111/j.1365-2427.2006.01706.x

Waltari, E., Schroeder, R., Mcdonald, K., Anderson, R. P., & Carnaval, A. (2014). Bioclimatic variables derived from remote sensing: Assessment and application for species distribution modelling. *Methods in Ecology and Evolution*, *5*(10), 1033–1042. https://doi.org/10.1111/2041-210X.12264

Wassila, M., Thiaw, P. H., Kamsu-Tamo, and E. B. (2018). The Floods in Equatorial East Africa during MAM 2018 Rainfall Season. *Climate Prediction Center, NOAA/NWS/NCEP*, (43rd NOAA Annual Climate Diagnostics and Prediction Workshop).

Waszkowiak, A., Frąckowiak, D., Wiktorowicz, K., & Miyake, J. (2002). A Brief Tutorial on Maxent. *Acta Biochimica Polonica*, *49*(3), 633–641.

https://doi.org/10.18388/abp.2002_3772

Wickham, H. (2016). (2016). *ggplot2: elegant graphics for data analysis.* Springer.

Wurtsbaugh, W. A., Paerl, H. W., & Dodds, W. K. (2019). Nutrients, eutrophication and harmful algal blooms along the freshwater to marine continuum. *Wiley Interdisciplinary Reviews: Water*, *6*(5), 1–27. https://doi.org/10.1002/wat2.1373

Zare Chahouki, M. A., & Piri Sahragard, H. (2016). Maxent modelling for distribution of plant species habitats of rangelands (Iran). *Polish Journal of Ecology*, *64*(4), 453–467. https://doi.org/10.3161/15052249PJE2016.64.4.-002

Chapter 5: Conclusion and Recommendations

This dissertation promotes the notion that wetlands are integral to our watersheds, an important component of the landscape, and play an instrumental role at the political, socioeconomic and ecological interface. This study stems from a hybrid of two frameworks that acknowledge the interlinkage of such a variety of sectors, Integrated Watershed Management (IWM) and Adaptive Management (AM). As suggested by previous studies (Overdevest et al., 2004; Wortley, Hero, & Howes, 2013), these frameworks provide an underpinning and holistic platform from which to evaluate the performance of policies and actions on the ground in relation to wetland management. The adaptive nature of these frameworks stresses the need for ecological monitoring and emphasizes that monitoring can help pinpoint the impacts of specific management practice on the environment, and can be particularly valuable for wetlands management (Leemhuis et al., 2016; Pahl-Wostl, 2007). To be truly effective for wetlands management, AM should not only include ecological monitoring data to inform all spheres involved: ecological, political and social economic spheres, but should also be informed by these spheres as part of the monitoring process (Swyngedouw, 2009).

The main objective of this dissertation was therefore to develop an ecological monitoring tool for freshwater ecosystems in Rwanda. This tool is referred to here as the odonate-based index, as it is based on odonates, insects that are biological indicators of habitat degradation and pollution (Miguel et al., 2017;Simaika & Samways, 2009). This tool was meant to enable deeper investigation of the extent to which landscape change, as shaped by political and socio-economic drivers, affects freshwater habitats in Rwanda. Additionally, by using odonates, which are charismatic insects (Simaika & Samways, 2018), the tool may engage and promote Citizen-Based Monitoring (CBM), ultimately instilling pro-environmental attitudes within local communities and setting the stage for collaboration between stakeholders

(Keough et al., 2006), as highlighted by IWM and AM. In this respect, the following series of questions were explored:

- What are the existing shortcomings of freshwater management in Rwanda, what are political and socioeconomic motives behind degradation of wetlands in Rwanda and how much are the IWM and AM principles included in the policies and laws that govern the environmental sector in Rwanda?
- What are the known odonate species in Rwanda, how are they distributed across the country, how does their abundance differ between ecological zones and seasons and to what extent does the odonate-based index reflect the habitat integrity, what are habitats that need special attention for conservation?
- How effective does the odonate-based index indicate the impact of the major socio-economic related threats, such as agriculture and mining, in a changing climate?

Through Integrated Watershed Management (IWM) and Adaptive Management (AM) lenses, this dissertation points out gaps that could lead to unfounded planning and the establishment of unachievable goals, which are significant shortcomings to the successful management of wetlands in Rwanda. The identified gaps include limitations in the full inclusion of all necessary stakeholders and integration of adaptive principles. This dissertation highlights ecological monitoring as a key to not only adaptive management but also community engagement, thus facilitating inclusive integration (Uyizeye et al., in prep.-a).

Given the importance of ecological monitoring that has been highlighted, this dissertation developed an odonate-based index tailored to Rwanda's ecosystems and socio-economy. This index provides ecologists, environmental decision makers and local communities with a robust monitoring tool for assessing freshwater habitats and a method to prioritize sites for conservation and restoration. The checklist of odonates known in Rwanda is determined to be 114 species (Uyizeye et al., in prep.-b). The abundance of these species was found to be significantly different between rainy and dry seasons as well as between ecological zones. I also highlight benchmark sites for each ecological zone that can play a reference role in restoration effort based on odonate-based index. Additionally, a list of hotspot habitats for odonates was provided. This is based on sites that harbor either unique species or high species richness. These are considered habitats that need special conservation attention (Uyizeye et al., in prep.-b).

The odonate-based index presented in this dissertation is useful, particularly, in the context of developing countries, given it is not only an effective bioindicator but also efficient in time and cost (Mangadze et al., 2019; Mendes et al., 2017; Parmar et al., 2016**)**. The data collection for odonate based monitroing requires as simple equipment as a hand book, sweep net, hand lens, bioboculars, and note book or an app on telephone. The effectiveness of this index is proven by its correlation with habitat integrity (Uyizeye et al., in prep.-b) as well as its ability to detect impacts from agriculture and mining, the major economy-driven threats to freshwater ecosystems in Rwanda (Uyizeye et al., in prep.-c). Additionally, an odonate-based tool can be used to not only monitor impacts caused by agriculture, mining and urbanization, but it can also serve as a means to monitor the effects of climate change. Climate variations are known to influence variables that also strongly influence odonate occurrence, such as bioclimatic and hydrological, precipitation of the coldest quarter and flow accumulation (Uyizeye et al., in prep.-c).

Recommendations

I propose the inclusion of an odonate-based tool in all ecosystem management programs, as well as monitoring protocols in Rwanda. These include environmental impact assessments, restoration programs and prioritization programs for the identification of sites needing special attention. When designing monitoring plans that use odonates, I recommend that one must account for differences in seasonality and ecological zones. This means that the comparison of localities should take place within the same season, especially when it is not feasible to sample in all seasons. Also, comparisons are more effective if the localities in question are within the same ecological zone. The consideration of season and ecological zone applies while monitoring single localities as well. Finally, given the strong interconnection and transboundary nature of freshwater systems in Africa, I recommend the development of similar tools tailored to other African regions (using their local odonate species), so that the odonate-based tool becomes a standard monitoring technique synchronized for effective management of freshwater ecosystems throughout the region and the continent.

References

Keough, H. L., & Blahna, D. J. (2006). Achieving integrative, collaborative ecosystem management. *Conservation Biology*, *20*(5), 1373–1382.

Leemhuis, C., Amler, E., Diekkrüger, B., Gabiri, G., & Näschen, K. (2016). East African wetland-catchment data base for sustainable wetland management. *Proceedings of the International Association of Hydrological Sciences*, *374*, 123–128. https://doi.org/10.5194/piahs-374-123-2016

Mangadze, T., Dalu, T., & William Froneman, P. (2019). Biological monitoring in southern Africa: A review of the current status, challenges and future prospects. *Science of the Total Environment*, *648*, 1492–1499. https://doi.org/10.1016/j.scitotenv.2018.08.252

Mendes, T. P., Oliveira-Junior, J. M. B., Cabette, H. S. R., Batista, J. D., & Juen, L. (2017). Congruence and the Biomonitoring of Aquatic Ecosystems: Are Odonate Larvae or Adults the Most Effective for the Evaluation of Impacts. *Neotropical Entomology*, *46*(6), 631–641. https://doi.org/10.1007/s13744-017-0503-5

Miguel, T. B., Oliveira-Junior, J. M. B., Ligeiro, R., & Juen, L. (2017). Odonata (Insecta) as a tool for the biomonitoring of environmental quality. *Ecological Indicators*, *81*(June), 555–566. https://doi.org/10.1016/j.ecolind.2017.06.010

Overdevest, C., Orr, C. H., & Stepenuck, K. (2004). Volunteer stream monitoring and local participation in natural resource issues. *Human Ecology Review*, *11*(2), 177–185. https://doi.org/10.1111/fme.12001

Pahl-Wostl, C. (2007). Transitions towards adaptive management of water facing climate and global change. *Water Resources Management*, *21*(1), 49–62.

https://doi.org/10.1007/s11269-006-9040-4

Parmar, T. K., Rawtani, D., & Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. *Frontiers in Life Science*, *9*(2), 110–118. https://doi.org/10.1080/21553769.2016.1162753

Simaika, J. P., & Samways, M. J. (2009). An easy-to-use index of ecological integrity for prioritizing freshwater sites and for assessing habitat quality. *Biodiversity and Conservation*, *18*(5), 1171–1185. https://doi.org/10.1007/s10531-008-9484-3

Simaika, J. P., & Samways, M. J. (2018). Insect conservation psychology. *Journal of Insect Conservation*, *22*(3), 635-642. https://doi.org/10.1007/s10841-018-0047-y

Swyngedouw, E. (2009). The political economy and political ecology of the hydro-social cycle. *Journal of Contemporary Water Research & Education*, *142*(1), 56–60.

Uyizeye, E., Kaplin A.B., Willey, L., Clausnitzer, V. (n.d.-a). Linking policy to practice through integrated and adaptive management of wetlands in Rwanda. *Chapter 2 of Doctoral Dissertation (Working Paper in Press)*, *Antioch Un*.

Uyizeye, E., Kaplin A.B., Willey, L., Clausnitzer, V., Kipping J.,& Dijkstra, K. D. B. (n.d.-b). Developing an odonate-based tool for monitoring freshwater ecosystems in Rwanda. *Chapter 3 of Doctoral Dissertation (Working Paper in Press)*, *Antioch Un*.

Uyizeye, E., Kaplin A.B., Willey, L., Clausnitzer, V., Kipping J.,& Dijkstra, K. D. B. (n.d.-c). The application of odonates as indicators for monitoring freshwater habitat integrity. *Chapter 4 of Doctoal Dissertation (Working Paper in Press)*, *Antioch Un*.

Wortley, L., Hero, J. M., & Howes, M. (2013). Evaluating ecological restoration success: A review of the literature. *Restoration Ecology*, *21*(5), 537–543. https://doi.org/10.1111/rec.12028