Use of invertebrates to evaluate the threat of water resource use on the wellbeing of selected river dominated estuaries in KwaZulu-Natal, South Africa

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ABSTRACT

Globally, estuaries are among the most productive ecosystems and have high ecological value as they provide suitable nursery habitats for many marine species. In contrast to the inshore marine environment which exhibits lower levels of food availability, heavy wave action and possible strong currents, estuaries benefit estuarine organisms through provision of appropriate conditions needed for growth including high food availability, sheltered habitat type and suitable water temperature. Estuarine systems also have high socio-ecological value as they provide goods and services such as water, sand, pollution control and fish for people. In South Africa, there are 258 functional estuaries, and of these, 23% are permanently open estuaries while only 2% are river mouths. Estuaries are vulnerable to external perturbations and these systems are highly threatened by anthropogenic impacts. The ever increasing human populations around estuarine systems introduce more stressors and make these systems more susceptible to human impacts. Examples of anthropogenic impacts acting on the estuarine ecosystems include effluent discharge, nutrient enrichment, introduction of invasive species, overfishing, development of harbours and jetties and water abstraction.

Many estuaries in the North Coast of KwaZulu-Natal (KZN), South Africa, are threatened by poor water quality, reduced flows and habitat alterations originating from anthropogenic land use activities. Although these estuarine systems are identified as significant and sensitive ecological assets, the mitigation opportunities for these are limited, mainly because the way in which they respond to the anthropogenic impacts is poorly understood. The uMvoti and Thukela estuaries, KZN, are impacted by the anthropogenic activities taking place in their catchments and they are susceptible to external stressors from land based sources. Consequently, these systems are considered to be highly threatened with impaired ecological functions including their ability to act as nursery grounds. Known anthropogenic impacts acting on these systems include habitat destruction, effluent impacts, nutrient enrichment, water abstraction and overfishing. In contrast the aMatikulu Estuary is considered less impacted with little anthropogenic impact upstream and this system is included in a nature reserve controlled by Ezemvelo KZN Wildlife.

Risk assessment is a process of assigning magnitudes and probabilities to the adverse effects of anthropogenic activities or natural catastrophes. Ecological Risk Assessment (EcoRA) is an organised approach that explains, describes and organises scientific facts, laws and relationships so as to provide sound foundation to develop adequate protection measures for the environment, which facilitates the establishment of utilisation strategies of the environment. Therefore, Ecological Risk Assessment evaluates the likelihood that adverse effects may occur or are occurring as a result of exposure to one or more stressors. A regional-scale ecological risk assessment is a summary of the complex interactions and effects of chemical and non-chemical stressors on the ecological endpoints. This method is an extension of traditional risk assessment methodology which incorporates multiple stressors, historical events, spatial structures and multiple endpoints in the assessment. A Regional Scale Risk Assessment using the Relative Risk Model (RRM) is a form of EcoRA that is carried out on a spatial scale where considerations of multiple sources of multiple stressors affecting multiple endpoints are allowed. Allowance for the landscape characteristics that may affect the risk estimate is made possible in this method. Therefore, the RRM method allows for the evaluation of multiple stressors being derived from multiple sources and impacting on a variety of species in a variety of habitats and a variety of locations. The RRM can be a useful approach that can contribute towards management of surface aquatic ecosystems in South Africa for the protection of biodiversity while allowing for the social and economic needs of the society. In this study we carried out a Regional Scale Risk assessment incorporating Bayesian Network probability modelling techniques to evaluate the threat of upstream (catchment) and local (Ilembe district municipal boundary) land use activities as well as threat of flow alterations to the ecological and selected social objectives/endpoints of the uMvoti, Thukela and aMatikulu estuaries. Macrozoobenthos and zooplankton samples were collected in the uMvoti, Thukela and aMatikulu estuaries from August 2014 to September 2016 using standard methods. Three sites were sampled in uMvoti and Thukela estuaries and four sites in aMatikulu/Nyoni Estuary. In the laboratory, samples were identified to the lowest taxa possible and enumerated. Bayesian Network relative risk models were developed using Netica by Norsys Software Corp.

Results from the macrozoobenthos assessment showed that the Thukela Estuary had the highest number of taxa (n = 24) followed by the aMatikulu/Nyoni (n = 11) and then uMvoti Estuary (n = 8). However, the aMatikulu/Nyoni Estuary displayed the highest abundance (31 764

no·m^{-2}) when compared with other estuaries studied. Following aMatikulu/Nyoni Estuary, the Thukela Estuary displayed abundance of (29 589 no·m^{-2}) followed by the uMvoti Estuary (10 336 no·m^{-2}). Results of the zooplankton assessment showed that the Thukela and aMatikulu/Nyoni estuaries had higher numbers of taxa (n = 10) when compared with the uMvoti Estuary (n = 5). The aMatikulu/Nyoni Estuary exhibited highest zooplankton abundance (15086.9 ind. m^{-3}) followed by Thukela (955.3 ind. m^{-3}) and then uMvoti Estuary (456.5 ind. m^{-3}). From the risk analyses, the scenario that displayed highest risk scores to endpoints was scenario 4 which predicted risk scores to each endpoint for year 2025 if no laws and management measures are implemented with biodiversity habitat displaying the highest risk scores. Scenario 3, a scenario before major resource development in the study area, had the lowest scores of risk for all the endpoints with the Thukela Estuary displaying the lowest of the scores for this scenario. The summation of all risk scores across all scenarios revealed that all selected endpoints were of highest risk in the uMvoti Estuary followed by the aMatikulu/Nyoni Estuary and then the Thukela Estuary. The safe environment endpoint was most likely to be affected in the uMvoti Estuary while biodiversity habitat was most likely to be affected in the Thukela Estuary. In the aMatikulu Estuary, productivity and safe environment were most likely to be affected. In the uMvoti and Thukela estuaries, higher risk was associated with social endpoints indicating that humans are at greater risk than the ecological components of these systems. However, in the aMatikulu/Nyoni Estuary, the ecological components of this systems were at a greater risk than humans as ecological endpoints displayed higher risk than social endpoints.

The results of the current study showed that the uMvoti Estuary was heavily impacted by the anthropogenic land use activities taking place in its upstream catchment. The impacts were reflected in the water quality of this system as well as biological communities. Such alterations were further witnessed in the risk analysis component of this study where all the endpoints displayed the highest risk in this system when compared with other estuaries studied. The Thukela Estuary is also impacted by anthropogenic land use activities taking place upstream and these impacts were reflected in its water quality and its biological communities. In contrast, the aMatikulu Estuary was in a relatively good ecological state when compared with the uMvoti and Thukela estuaries in terms of zooplankton and macrozoobenthos abundance and species richness although this system is likely to be at increased risk and thus need urgent management.

PREFACE

The data described in this thesis were collected in KwaZulu-Natal, Republic of South Africa from August 2014 to September 2016. Laboratory work was carried out while registered at the School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, under the supervision of Dr. Gordon O'Brien, Prof Victor Wepener, and Prof Colleen T. Downs.

This thesis, submitted for the degree of Doctor of Philosophy in Science in the College of Agriculture, Engineering and Science, University of KwaZulu-Natal, School of Life Sciences, Pietermaritzburg campus, represents original work by the author and has not otherwise been submitted in any form for any degree or diploma to any University. Where use has been made of the work of others, it is duly acknowledged in the text.

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Madonna Vezi
May 2017

I certify that the above statement is correct and as the candidate's supervisor I have approved this thesis for submission.

……………………………..

Professor Colleen T. Downs

Supervisor

May 2017

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DECLARATION 1 - PLAGIARISM

I, Madonna Vezi, declare that

1. The research reported in this thesis, except where otherwise indicated, is my original research.

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DECLARATION 2 - PUBLICATIONS

DETAILS OF CONTRIBUTION TO PUBLICATIONS that form part and/or include research presented in this thesis:

Publication 1 (Formatted for submission to Marine and Freshwater Ecosystems)

M Vezi, CT Downs, V Wepener & G O'Brien

Macro-benthic invertebrate communities in selected river dominated estuaries in KwaZulu-Natal, South Africa: effects of altered water quality and quantity

Author contributions:

MV conceived paper with CTD and GO. MV collected the data and analysed the data with support from CTD, GO and VW. MV wrote the manuscript. CTD, VW and GO edited and contributed valuable comments to the manuscript.

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M Vezi, CT Downs, V Wepener & G O'Brien

Response of zooplankton communities to altered water quality and quantity in selected river dominated estuaries in KwaZulu-Natal, South Africa

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M Vezi, CT Downs, V Wepener & G O'Brien

Application of Relative Risk Model for evaluation of ecological risk in selected river dominated estuaries in KwaZulu-Natal, South Africa.

Author contributions:

MV conceived paper with CTD and GO. MV collected the data and analysed the data with support from CTD, GO and VW. MV wrote the manuscript. CTD, VW and GO edited and contributed valuable comments to the manuscript.

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May 2017

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CHAPTER 1

Introduction

1.1 Definition of an estuary and estuarine characteristics

Estuaries are among the world's most productive ecosystems and they are of economic and ecological value (Chuwen et al., 2009a; McLusky, 2004; Vasconcelos et al., 2010). These systems are characterized by continuously changing mixture of saline and fresh waters as well as fine and course sediments from the river and the sea respectively. The size of particles and speed of currents control the distribution of sediments in estuarine systems (Day, 1981a; Levin et al., 2001; McLusky, 2004). These systems are characterized by horizontal and vertical salinity gradients (Louw, 2007; McLusky, 1993). Freshwater inflow, levels of tidal mixing as well as local topography are the drivers of the degree of such salinity gradients (Boaden and Seed, 1985; Louw, 2007). There is variation in temperature regimes with depth, marine and continental climate as well as the input of water from adjacent systems with varying temperatures (McLusky, 2004). The above mentioned characteristics contribute to an unstable environment causing organisms to deal with variability in habitat (e.g. sediment composition and distribution) and physiological stress (Harrison and Whitfield, 2006; James and Harrisson, 2009; MClusky 1999; Perillo, 1995). Consequently, relatively few species have adapted to thrive in these systems (Levin et al., 2001). Therefore, estuaries generally have low species richness when compared with the adjacent marine and freshwater environments although species often occur in high densities (Elliott and McLusky, 2002; Levin et al., 2001; McLusky, 2004).

"An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage" (Pritchard, 1967). Smaller temporarily open/closed estuaries and lagoons were not accounted for in this definition as it was essentially based on features of large northern hemisphere estuaries. Day (1980, 1981a) revised Prichard's (1967) definition to: "An estuary is a partially enclosed coastal body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of sea water with fresh water derived from land drainage''. Fairbridge (1980) also proposed a definition of an estuary as, "An estuary is an inlet of the sea reaching into a river valley as far as the upper limit of tidal rise, usually being divisible into three sectors: (a) a marine or lower estuary, in free connection with the open sea; (b) a middle estuary subject to strong salt and freshwater mixing; and (c) an upper or fluvial estuary, characterised by freshwater but subject to strong tidal action. The limits between these sectors are variable and subject to constant changes in the river discharges".

Perillo (1995) and Elliott and McLusky (2002) argued that in Day's (1980, 1981a) definition, tidal variation was omitted and emphasis was on salinity. Tides play a significant role by providing energy for the mixing mechanism in estuaries but wind can sometimes have a considerable effect (Perillo, 1995). Tidal mixing does not only have impact on salinity in these systems, it also influences processes such as erosion and circulation. The tidal action causes some changes to river discharge, pollutants and sediment transport characteristics in the fluvial reaches (Mclusky 1999; Perillo, 1995). After Perillo's (1995) argument and revision he proposed a new definition as, "An estuary is a semi-enclosed coastal body of water that extends to the effective limit of tidal influence, within which seawater entering from one or more free connections with the open sea, or any other saline coastal body of water, is significantly diluted with fresh water derived from land drainage, and can sustain euryhaline biological species for either part or the whole of their life cycle". He further stated that this definition includes aspects which were omitted before i.e. (i) hierarchical estuaries that possess primary to tertiary tributaries such as the Chesapeake Bay, (ii) the existence of more than one free connection, hence coastal lagoons are also included in the definition, (iii) the coexistence of tidal action and invasion of sea water and (iv) the inclusion of biological aspects where the estuary can be a habitat for species that can tolerate a wide range of salinities.

The Water Framework Directive (WFD) of the European Union regards an estuary as an isolated habitat but habitats like salt marsh, reedbeds, sand and mud flats are also included (Elliott and McLusky, 2002). Romao (1996) then defined an estuary as, "Downstream part of a river valley, subject to the tide and extending from the limit of brackish waters. River estuaries are coastal inlets where, unlike large shallow inlets and bays' there is generally a substantial freshwater influence. The mixing of freshwater and seawater and the reduced current flows in the shelter of the estuary lead to the deposition of fine sediments, often forming extensive intertidal sand and mud flats. Where the tidal currents are faster than the flood tides, most sediments deposit to form a delta at the mouth of an estuary". Although the above definition is somewhat long, Elliot and McLusky (2002) regarded this definition as more realistic and accurate and they further regarded it as closer to the definitions of Prichard (1967) and Fairbridge (1980) than any other succeeding definitions which have considered estuaries as the "non-tidal brackish seas" or "river plumes extending into open seas". The South African National Water Act 36 of 1998 gave definition of an estuary as, "a partially or fully enclosed body of water - (a) which is open to the sea permanently or periodically; (b) within which sea water can be diluted to an extent that is measurable, with freshwater drained from land".

In Day's (1980, 1981a) definition, hypersaline conditions were not accounted for. According to Potter et al. (2010), formation of sandbars at the mouths of estuaries and hypersaline conditions were excluded in the previous definition by Day (1980). Potter et al. (2010) modified Day's (1980) definition to, "An estuary is a partially enclosed coastal body of water that is either permanently or periodically open to the sea and which receives at least periodic discharge from a river(s), and thus, while its salinity is typically less than that of natural sea water and varies temporally and along its length, it can become hypersaline in regions when evaporative water loss is high and freshwater and tidal inputs are negligible". Pritchard's (1967) and Day's (1980) definitions have therefore been extended to include small, temporarily open/closed estuaries (TOCEs), which are the main dominant type of estuaries in South Africa.

1.2 Importance of estuaries and major anthropogenic impacts

Estuaries are of high ecological value as they provide suitable nursery habitats for many marine species (Beck et al., 2003; Elliott and McLusky, 2002; Nicolas et al., 2007; Potter and Hyndes, 1999; Vasconcelos et al., 2010). Estuarine ecosystems benefit these species through provision of appropriate conditions needed for growth. These include increased food availability, sheltered habitat type and suitable water temperature, which contrasts with the inshore marine environment exhibiting lower levels of food availability, heavy wave action and possible strong currents (Pittman and McAlpine, 2001; Vasconcelos et al., 2010; Wasserman and Strydom, 2011). These marine species remain in estuaries for part or for entire life cycle after which they may migrate back to the marine environment to join the adult population (Perillo, 1995; Strydom and Whitfield, 2000; Vasconcelos et al., 2010; Whitfield, 1999b). The survival of the early life stages for many marine species is dependent on estuaries as these systems provide important nursery areas (Whitfield, 1999b). Also, many bird species rely on estuaries for their food (Hockey and Turpie, 1999).

Estuarine systems act like filters as they trap excess nutrients coming from inland rivers (Scharler and Baird, 2005; Taljaard et al., 2009; Telesh and Khlebovich, 2010). These systems display relatively higher nutrient loads and organic matter production when compared with rivers and marine environments (de Villiers and Hodgson, 1999; Turpie et al., 2002). Nutrients in estuaries are retained and recycled as a result of the trapping properties of these ecosystems (Odum, 1969). Mixing is the important process for primary production in aquatic systems as it brings buried nutrients into the water column (Lewis, 1996). More nutrients are transported to the lower reaches of estuaries and adjacent marine environment during high river inflow (Carić et al., 2012). High amounts of sorbed nutrients in estuarine sediments are made possible by the sorptive capacity of fine clay particles (Carić et al., 2012). Tidal exports of nutrients from some estuaries may contribute to the coastal ocean productivity (Howarth, 1988; Levin et al., 2001). Estuaries are the productive systems accessible to humans for available proteins (e.g. fish) and recreational activities (Lamberth and Turpie, 2003). These estuarine systems are also used by nearby industries for disposal, processing and transportation of waste (Lauff, 1967; Onojake et al., 2011).

Estuaries are among the most threatened ecosystems especially because of increased anthropogenic impacts. These systems are vulnerable to external perturbations. Anthropogenic activities may lead to the alteration of ecological functioning of these systems including their capacity to function as nursery habitats (McLusky, 2004; Nicolas et al., 2007). The ever increasing human populations near estuarine systems introduces more stressors and makes these systems more susceptible to human impacts (Nicolas et al., 2007; Perissinotto et al., 2010; Thomas et al., 2005). Effluent discharges, nutrient enrichment, introduction of invasive species, development of harbours, causeways and marines, overfishing and water abstraction are the examples of anthropogenic impacts (Allanson and Read, 1995). Consequently, estuaries display changes in biological communities, hydrodynamics, and shifts in species diversity (Allanson and Read, 1995; Nicolas et al., 2007). Shifts in biological community compositions may also result from increased sediment load, and temperature changes in marine, estuarine and river waters as a result of global climate changes, causing species to alter their distribution in affected estuaries (James and Peterson, 2011). Alterations in estuarine morphology may result from changes in erosion and siltation (Pontee et al., 2004). Reduced flow has a negative effect on estuaries as these systems rely on freshwater inflow to open the inlet, and flush sediments, pollutants and nutrients from the estuary. Such reductions result in alterations of estuarine water quality (Pontee et al., 2004). River inflow, together with tidal exchange, enhances turbidity gradients in estuaries which are important during the nursery phase of some fish species through provision of olfactory cues for juveniles and reduction of predatory rates through impairment of visibility to predatory fish (Allanson and Read, 1995; Blaber and Blaber, 1980; Whitfield, 2005). Turbidity also increases the catchability of many fish species, predominantly the large individuals that move into the turbid waters in search of prey (Lamberth et al., 2009). Reduction in freshwater flow into an estuary may result in closure of the estuary mouth, particularly by a sand bar, for prolonged periods and this may limit the migration of marine species into the estuary and back to sea, thus negatively impacting species richness and population size in the marine environment (James and Peterson, 2011; Mann and Pradervand, 2007). In estuarine systems which supply high amounts of terrigenous sediment to the adjacent marine bank e.g. Thukela Estuary, reduced flows and changing catchment land use activities may alter the sediment load deposited into the marine environment (Lamberth et al., 2009). In the long-term, increased fine sediments would favour penaeid prawns and flatfish species, but would be harmful to filter-feeding invertebrates and eventually the reef-dwelling fish species that prey on them (Lamberth et al., 2009).

1.3 South African estuaries

There are approximately 255 functional estuaries in the coast of South Africa (Whitfield, 2000). These systems together with their percentage contribution have been categorized into five groups (Whitfield, 1999a; Whitfield, 2000). These are temporarily open/closed estuaries (TOCEs) (72 %), permanently open estuaries (POEs) (18%), estuarine lakes (3 %), river mouths (5 %), and estuarine bays (2 %) (Whitfield, 1992; Whitfield, 2000).

1.3.1 Permanently open estuaries

Permanently open estuaries (POEs) are permanently connected to the sea and have a moderate tidal prism which is typical of South African estuaries (Whitfield, 1992). These estuarine systems mostly dominate in the northern hemisphere particularly along the North American and European coastlines (Perissinotto et al., 2010; Potter et al., 2010). Such systems are generally characterized by high runoffs throughout the year and large catchments (> 500 km^2) (Whitfield, 1992; Whitfield and Bate, 2007). The principal drivers of the mixing processes in the water column of these systems are the tidal and river flows and normally their mean salinities fluctuate between 15 and 35 (Whitfield, 2005). Headwaters of POEs often exhibit oligohaline conditions while the mouth regions exhibit euryhaline conditions. Tidal mixing processes can become dominant if flows are reduced as a result of anthropogenic land use activities happening upstream (Allanson and Read, 1995). Permanently open estuaries with limited freshwater supply can sometimes experience hypersaline conditions (Whitfield, 2005), e.g. the Kromme Estuary hypersaline conditions were reported in the upper reaches (Wooldridge and Callahan, 2000), and the Kariega Estuary where salinities of 35 and 42 were measured in the lower and upper reaches respectively (Matcher et al., 2011). During periods of high river flow, oligohaline conditions can prevail in the lower and middle reaches of POEs (Whitfield, 2005). However, these events will become increasingly rarer due to the increased number of impoundments in river catchments which capture most of the flood waters for anthropogenic use (Whitfield, 2005). Reduced freshwater flow may result in elevated flood tidal deltas near the mouth region and this might further reduce the tidal exchange between the estuary and the sea (Cooper, 2001; Grange et al., 2000; Whitfield, 2005).

1.3.2 Temporarily open/closed estuaries

The majority of temporarily open/closed estuaries (TOCEs) in South African coast have small river catchments usually less than 500 km^2 (Whitfield, 1992; Whitfield, 2005). This group of estuaries also occurs in other parts of the world e.g. Australia (Coorong and Harbord estuaries and Smiths Lake) where TOCEs extend across South Eastern coastline of Australia, extending from South Australia through to Victoria, including the Tasmanian coastline (Roy et al., 2001), Uruguay, Brazil (Bonilla et al., 2005), India, Sri Lanka (Ranasinghe and Pattiaratchi, 1998; Ranasinghe and Pattiaratchi, 2003) and the USA e.g. in Texas and California (Gobler et al., 2005; Kraus et al., 2008). During low flow seasons, a sand bar forms at the mouth of these systems and they lose connection with the sea (Perissinotto et al., 2010; Whitfield et al., 2012). Following the high river inflow during the wet season, water levels in these systems rise and exceed the sand bar after which the estuary usually breaches (Froneman, 2002; Perissinotto et al., 2010; Whitfield, 1992; Whitfield, 2000). After the outlet has been formed, the water level in the estuary drops and exposes some areas of estuary bed. These areas may have been colonised by rich communities of fauna like macrozoobenthos and flora like microphytobenthos and macrophytes which were submerged for longer periods (Perissinotto et al., 2010; Whitfield, 1992). These systems experience short periods of tidal exchange after the estuary has emptied with re-formation of a sandbar resulting from reduced flow ending the open phase (Froneman, 2002; Perissinotto et al., 2010; Perissinotto et al., 2000).

Closed phase of TOCEs may take days, months or even years depending on the river flows (Perissinotto et al., 2010). Ecological functioning, physico-chemical processes, hydrodynamics and biological structure of TOCEs are controlled by this dynamic closing and opening process (Perissinotto et al., 2010). Water abstraction and impoundments have reduced freshwater inflow in many South African estuaries. This reduction of inflow has negatively impacted TOCEs as these systems rely on increased freshwater flow to open the estuary mouth (Grange et al., 2000). In contrast, unnatural elevated water supply from waste water treatment works has caused mouths of some South African TOCEs to open more frequently, and simultaneously also increased the nutrient supply e.g. the Mhlanga Estuary (Lawrie et al., 2010; Thomas et al., 2005). If low freshwater inflow persists, prolonged estuarine mouth closure may result and limit migration of invertebrates and fish between the sea and the estuary (Mann and Pradervand, 2007; Whitfield, 2005). If there is permanent closure of the mouth, marine species may become locally extinct and species richness will decline with the estuary dominated by freshwater and estuarine species and hypersaline conditions may prevail especially during periods of high evaporation rates (Whitfield 2005). Low flows reduce the amount of nutrients transported to the estuary which in turn reduces primary production which support zooplankton (Whitfield, 1995; Whitfield, 2005). Consequently, reduced zooplankton abundance together with occasional hypersaline conditions may reduce species diversity and abundance of zooplanktivorous fishes (Whitfield, 1995; Whitfield, 2005). Although the TOCEs are found in other parts of the world, they have received little attention relative to permanently open estuaries. However, TOCEs along the South African coast have received considerable attention (Froneman, 2001; Froneman, 2002; Froneman, 2004; Nozais et al., 2001; Perissinotto et al., 2000; Perissinotto et al., 2002; Perissinotto et al., 2003).

1.3.3 River mouths

General characteristics of South African river mouths include large catchment areas (> 10 000 km^2), permanently open mouths and high silt load derived from the land (Whitfield, 1992). Tidal prism of these systems is however relatively small (< 1 x 10^6 m^3) (Whitfield, 1992). River inflow has more influence in the physical processes of these systems when compared with the marine tidal influence, and saline waters rarely occur in the upper reaches during moderate to high freshwater runoff, although dilution of sea and freshwater may be apparent in the lower reaches during low flow (Whitfield, 1992). These river mouths are generally relatively shallow (< 2 m deep) irrespective of amount of water passing through them (Whitfield, 1992), however, a depth of 15 m can be measured following periodic floods (Swart et al., 1988). Freshwater inflow controls water temperature of these systems although marine influences can sometimes control water temperatures in the bottom waters of the lower reaches (Whitfield, 1992). These river mouths are generally dominated by freshwater organisms (Day, 1981b). Examples of typical river mouths in South Africa include the Mzimvubu, Storms River, uMvoti and Thukela estuaries (Whitfield, 2000).

1.3.4 Estuarine bays

Contrary to the river mouths, one characteristic of a South African estuarine bay is the continuous substitution of estuarine water with sea water in the lower portion of the channel (Whitfield, 1992). Tidal prism of these systems is generally large (>10 x 10^6 m^3) (Whitfield, 1992). Tides and wind are the most important drivers of mixing process in estuarine bays and there is prominent salinity stratification in the middle and upper reaches (Largier et al., 2000; Whitfield, 1992). Salinity levels in the lower reaches of these systems is usually greater than 25 e.g. the Knysna system (Largier et al., 2000), however, during high river flow salinity levels below this may be recorded in the lower reaches (Grindley, 1985; Whitfield, 1992). Along the South African coastline there are natural (e.g. the Knysna system) and artificial (e.g. Richards Bay and Durban Bay systems) estuarine bays (Whitfield, 1992; Whitfield, 2000).

1.3.5 Estuarine lakes

Most South African estuarine lakes are separated from the marine environment by vegetated sand dune systems (Whitfield, 1992). There are only eight estuarine lakes in the coast of South Africa and these are Mgobezeleni, St. Lucia, Kosi, Wilderness, Nhlabane, Kleinemonde, Klein, and Swartvlei estuarine systems (Whitfield, 2000). Some estuarine lakes get isolated from the sea for a couple of years after which they lose their estuarine characters and are then referred to as coastal lakes. Generally, remnant estuarine biota tolerant of freshwater conditions still inhabit these systems (Whitfield, 1992). Some systems have a temporal marine connection e.g. Swartvlei system, while others have a permanent marine connection e.g. Kosi system. Wind is the most important driver of the mixing process in the water column even in the deeper systems and hypersaline conditions may prevail in other systems during low flow or during drought e.g. St Lucia system (Vivier and Cyrus, 2009; Whitfield, 1992). St. Lucia is one example of an estuarine lake that receives about 50% of its input from precipitation (Vivier and Cyrus, 2009; Whitfield, 1992). Generally, solar heating is one important factor influencing water temperatures in these systems (Whitfield, 1992). From the five types of estuaries on the South African coast, the current study focuses on river mouth (uMvoti and Thukela) as well as permanently open (aMatikulu/Nyoni) estuarine systems.

1.4. Sensitivity of estuaries to reductions in river flows and water quality alterations

South African estuaries particularly have more complex dynamics because of tidal influence and sea water intrusion (DWAF, 1999). As mentioned earlier, estuaries are strongly dependent on extreme conditions, among the most crucial are river base flows, tidal exchange and major floods. Changes in base flow and major flood patterns can have significant effects on the functioning of an estuary (Hitchcock and Mitrovic, 2015). All estuaries are sensitive to reductions and changes in river flows (Pontee et al., 2004). Catchment size is not necessarily linked directly to the runoff due to the arid climate of South Africa (Whitfield and Bate, 2007). Permanently open systems generally have larger catchments with significant river flow occurring throughout the year, while TOCEs tend to have smaller catchments (often < 100 km^2) and are normally characterised by a strong seasonal variation in runoff (Whitfield, 1992). The Buffels, Spoeg and Groen catchments in the Northern Cape are examples of large arid catchments that drive mouth status through floods (Whitfield and Bate, 2007). These rivers only flow after substantial rainfall and episodic floods have been experienced (Heydorn and Grindley, 1981a; 1981b; 1981c). Breaching of their respective estuaries is mostly associated with flood events. Marine overwash of the sand bar might be important for a few days after mouth closure to modify salinity of the water column (Whitfield and Bate, 2007). If such estuaries were situated on a high-energy beach, they would tend to close quickly (Whitfield and Bate, 2007). There are certain parameters (particularly physical parameters) that can indicate if an estuary is sensitive to modifications or not. These parameters which are important indicators could be used to determine the extent to which estuaries would be sensitive to flow alterations e.g. frequency of mouth closure (DWAF, 2010).

For many South African estuaries, especially the smaller ones, the most important factor in keeping the mouth open is river flow, and particularly base flows. Other factors and / or combination that may contribute to estuary's sensitivity to mouth closure include size of an estuary. Generally, larger estuaries are less sensitive to mouth closure when compared with small estuaries. When larger estuaries breach they tend to scour deeper mouths due to higher outflows, which generally take longer to close, e.g. Bot and Klein estuaries (DWA, 2010). However, when the mouth of a large estuary closes, a large amount of water is needed to first fill up the estuary before breaching can occur, therefore substantial river inflow is needed to ensure breaching in large compared with smaller estuaries (DWA, 2010). Small estuaries are vulnerable to reduced flows as they rely on freshwater inflow to open the mouth, transport sediment, nutrients and pollutants. Tidal flow can maintain an open mouth state in larger temporarily open estuaries (>150 ha) when runoff decreases during the low flow season (Whitfield and Bate, 2007). This is the case in permanently open estuaries, estuarine bays and some river mouths (Whitfield and Bate, 2007). The only exception to this rule are estuarine lakes because they close despite their significant size due to possible factors like extended low flow, high evaporation rates, high sediment availability and high wave energy (Whitfield and Bate, 2007).

Another component that contributes to South African estuary's sensitivity to mouth closure is availability of sediment (DWA, 2010). In general, the larger the amount of sediment available in the adjacent marine environment, the greater the sensitivity to mouth closure, e.g. most estuaries along the KZN coastline. In estuaries where there is relatively little sediment available, for example on a rocky coastline or where longshore transport is further offshore, e.g. Nahoon Estuary, Eastern Cape Province, the system would be less sensitive to flow reductions (DWA, 2010). When there is erosion in the river catchment the estuary is likely to experience sedimentation and when the flows are reduced these sediments accumulate near the mouth area with not enough water to scour them to the sea. Nearshore habitats which are continuously eroded by oceanic currents are replenished by sediment export which in turn provide a refuge for many fish by increasing turbidity (Cyrus and Blaber, 1992).

Wave action in the mouth is another factor that contributes to an estuary's sensitivity to mouth closure (DWA, 2010). In general, the stronger the wave action in the mouth the greater the sensitivity to mouth closure. The wave conditions near the mouth area are influenced by the degree of protection to the mouth e.g. by a beach slope. A steep beach slope indicates that high energy wave action occurs on the beach at the mouth, resulting in higher suspended sediment load (DWA, 2010). This type of beach slope is characteristic of the KZN coastline (DWA, 2010). Generally, the steeper the slope of a beach, the higher the suspended sediment load in the mouth area, therefore the greater the sensitivity to mouth closure. A shallow beach slope indicates that less energetic wave action takes place at the mouth and therefore provides a special type of protection against wave action (DWA, 2010).

Freshwater is an essential factor controlling biological productivity of estuaries and coastal areas globally including South Africa. Because of economic development and population growth which has resulted in high demand for water resources, there has been a reduction in water discharge of many rivers worldwide as a result of damming/ impoundments and water abstraction (Zhang et al., 2012). Globally there has been a construction of numerous artificial dams and other hydraulic structures to impound water for the purpose of drinking water supply, hydropower production, irrigation, flood control and navigation (Gleick et al., 2006). Riverine input has long been identified as one of the most important factors controlling the functioning of South African estuaries (Day et al., 1954). Freshwater flow can play a significant role in elevating nutrient concentrations in these estuaries (Eyre and Ferguson, 2006). Again, flow events change water residence time which in turn can have negative impacts on phytoplankton productivity. The existence of dams has significant environmental impacts on the downstream estuarine waters (Jeong et al., 2014). When flows are reduced as a result of anthropogenic activities taking place upstream, the salinity levels in the receiving estuary may rise to the detrimental levels that may affect estuarine communities (Havens, 2015). Decrease in freshwater flow may result in an incursion of saline water in the upper reaches of an estuary, exposing organisms here to salinity stress (Attrill et al., 1996). Studies have been conducted on the tolerance of marine invertebrates to low salinities (e.g. Guerin and Stickle, 1992), however little is known about the tolerance of freshwater invertebrates to elevated salinities or the resulting patterns of distribution over the river/estuary interface (Attrill et al., 1996, Kefford et al., 2016).

Examples of physiological stress resulting from increasing salinity may include die-off as well as reduced growth of sea grass (Zieman et al., 1999) and oyster mortalities (Petes et al., 2012). When salinity increases beyond optimal levels and beyond levels that fauna can tolerate, sedentary species especially may decline in numbers. Some species may be able to migrate further upstream to reach regions where the salinity levels are tolerable (Havens, 2015). Regular reduced environmental flows may cause degraded estuarine environments and the loss of important nursery habitats for fish and shellfish resources (Powell and Matsumoto, 1994). Reduced environmental flows also affect commercial landings of fish and shell fish (Havens, 2015). Previous studies have shown a correlation between freshwater inflow and catches of estuarine fish and crustaceans (Robins et al., 2005). This suggests that freshwater from the river may be providing nutrients and carbon to enhance productivity in estuaries (Robins et al., 2005). It is expected that with climate change and increased frequency of droughts, production of commercially important fish species will decline (Havens, 2015). Higher frequency of extreme conditions will change both species composition and patterns of energy flows (Havens, 2015).

Estuarine ecosystems support approximately 75% of the world's human population and the number of residents living in coastal areas is continuously rising (Vitousek et al., 1997). As mentioned earlier these estuarine ecosystems receive high concentrations of land based nutrients and other pollutants through surface runoff, atmospheric deposition and groundwater discharge, much of it transported via rivers draining urban centers and agricultural watersheds (Howarth et al., 1996; Jaworski et al., 1997; Paerl et al., 2002). At smaller spatial scale, riparian forests and wetlands are likely to lower the effects of agricultural and urban land use (Osborne and Kovacic, 1993). The main characteristics of aquatic biotic communities associated with high agricultural and urban land use include lower species diversity, less trophic complexity, altered food webs, altered community composition and reduced habitat diversity (Correl, 1997; Roth et al., 1996). Anthropogenic nutrient inputs have increased significantly over the past few decades (Howarth et al., 1996, Galloway et al., 2008).

Many estuaries are facing nutrient-over-enrichment in the form of nutrient enhanced primary production and these changes in nutrient availability result in increased eutrophication which is a growing threat facing coastal ecosystems (Bricker et al., 2008). Unused or partially degraded organic matter settles to the sediments and serves as a fuel for microbial decomposition converting organic matter to carbon dioxide and inorganic nutrients, an oxygen demanding process (Paerl, 2006). Detritus may serve as a direct food source for detritivorous fish and invertebrates (Blaber, 2000; Whitfield, 1998). Waters possessing high amounts of readily degraded organic matter tend to consume high amounts of oxygen. The imbalance between relatively high rates of oxygen consumption and low rates of oxygen re-supply may cause a drop in dissolved oxygen content to levels that may be detrimental to aquatic plants and animals (Paerl, 2006). When estuarine waters reach dissolved oxygen content of less than 4 $mg \cdot l^{-1}$ they are referred to as hypoxic and are commonly stressful to organisms of higher trophic level (Paerl, 2006). When oxygen concentrations in water are not detectable those waters are said to be anoxic and are fatal to finfish, shellfish and invertebrates (Diaz and Rosenberg, 1995; Pihl et al., 1991; Renauld, 1986).

Apart from making coastal environments undesirable for human use and threatening commercial harvests, eutrophication also significantly restructures natural communities and ecosystem functioning (Fox et al., 2009; Oesterling and Pihl, 2001; Valiela et al., 1997). As mentioned anthropogenic activities such as sewage effluent discharges have effects on estuarine systems (Lawrie et al., 2010). There has been a widespread of eutrophication in rivers and estuaries of South Africa and other parts of the world as a result of anthropogenic activities and this has caused significant degradation of many estuarine systems (Conley et al., 2009; Walmsley, 2000). In the majority of environments, industries are the main sources of pollution (Kanu and Achi, 2011). River systems and ultimately estuaries are the primary means for waste disposal, mostly effluents from industries located near them. These industrial effluents pollute water bodies and subsequently alter the chemical, physical and biological nature of the receiving systems (Sangodoyin, 1995). High levels of pollutants in aquatic systems elevate levels of biological oxygen demand (BOD), chemical oxygen demand (COD), total dissolved solids (TDS), total suspended solids (TSS), toxic metals such as Cd, Cr, Ni and Pb, as well as counts of fecal coliform. Such alteration makes water unsuitable for drinking, irrigation and aquatic life (Kanu and Achi, 2011). High biochemical oxygen demand originates from biodegradable wastes such as those from human sewage, pulp and paper industries, slaughter houses and chemical industry (Kanu and Achi, 2011). Other such sources may include plating shops and textiles, and these may be toxic and may require on-site physicochemical pre-treatment before discharge into municipal sewage system (Emongor et al., 2005). Exposure to pulp and paper effluent may result in a variety variety of alterations in some fish physiological processes including impaired reproduction, pathological lesion, growth disturbances, and biochemical responses (Owens, 1991). Among pathological changes associated with fish exposed to pulp and paper effluent, Owens (1991) and Everall (1992) mentioned increased prevalence of fin erosion and vertebral deformities, altered resistance to infectious diseases, neoplasia and hemolytic anemia. It is thus important to have tools to evaluate the adequacy of pulp and paper effluent outputs and the regulations governing these to protect fish and other biotic components as well as the habitat. Such tools are of great importance since they can provide useful and traceable information on possible alterations of life history characteristics. Hazard assessment of pulp and paper effluents in aquatic environments is a complex task (Owens, 1991). Factors that have been reported to hinder hazard assessment include variety of individual compounds in pulping effluent and site-specific differences in processes, effluent treatment, and receiving ecosystems (Owens, 1991). Hazard assessment of pulping effluents require multidisciplinary efforts that integrate chemical, toxicological and biological data. Mutagenic and cancer promoting agents have been measured several times in pulp mill effluents (Owens, 1991). Increased prevalence of cutaneous pigment cell tumors were documented in fish collected nearby a bleached mill (BKM) in Germany (Owens, 1991). These results were also produced experimentally in fish treated with effluent (Kinae et al., 1990). Liver tumors have also been reported in fishes exposed to the pulp effluents (Metcalfe et al., 1995).

1.5. Protection of water resources and existing water resource protection legislation

The primary sources of water for agricultural, industrial and domestic use in South Africa are river systems and they supply more than 85% of all the water used in South Africa with the groundwater system providing the remainder (Ashton, 2007). One characteristic of South Africa's climate is an uneven, poorly predictable and highly seasonal distribution of rainfall, moreover, potential evapotranspiration rates may exceed rainfall (Tyson, 1987). Droughts are a common event and are often followed by equally extreme floods (Tyson, 1987) which are likely to increase with global climate change predictions. To increase the reliability of supplies for South African urban and rural users, multiple water supply reservoirs/ impoundments, farm dams and inter-basin transfer schemes have been constructed (Ashton, 2007).

The majority of South Africa's rivers are impacted negatively by discharges of treated domestic and industrial effluent from towns and cities, and with return flows from irrigated agriculture contributing agrochemicals (Ashton, 2007). The ever increasing human population size coupled with increased efforts to meet increasing demands for food, fibre, fuel and freshwater place heavy demands on the country's limited resources (Ashton, 2002).

Introduction of alien species, destruction of wetlands, removal of riparian vegetation and disruption of connectivity between freshwater habitats are known to accentuate these threats and effects further (Roux et al., 2002).

The ecological state of South Africa’s surface water is deteriorating continuously causing a decline in provision of key ecosystem services (Ashton, 2007; Driver et al., 2005). At the beginning of the current century only about 30 % of South Africa’s main rivers were still natural and sustainable while 47 % were modified to varying degrees and 23% were irreversibly transformed (Nel et al., 2004). It is acknowledged that the conservation goals required to maintain the aquatic biodiversity of surface aquatic ecosystems are currently unattainable in South Africa as a result of excessive use of services of these ecosystems (O'Keeffe, 1989, Rivers-Moore, 2008). The effective conservation of a river system requires careful management of the entire catchment for the achievement of sustainable social, economic and ecological objectives (Chan et al., 2006; Gilman et al., 2004).

In order to establish integrated management plans for surface aquatic ecosystems in South Africa there should be a close engagement of stakeholders in the social and institutional decision making processes (DWAF 2006). In South Africa, the use and protection of all aquatic ecosystems must be undertaken within a legislative context. Ashton (2007) stated that the establishment of management plans should allow for the balance between the adequate protection of terrestrial and aquatic biodiversity while concurrently allowing for the social and economic needs of the society. This involves the development of integrated management plans that have a wide range of conservation and use objectives for the specific river systems that are affected by multiple stressors with consideration of the unique characteristics of the ecosystem (O'Brien, 2012). In South Africa, formal custodianship of water resources is vested in the Department of Water and Sanitation (DWS) (DWAF, 1996) but several government departments (e.g. Department of Environmental Affairs) and sectors of government (national, provincial and local) are also responsible for different aspects of the use and management of water resources. Through the DWS, the National Water Resource Strategy describes how the water resources of South Africa should be used, protected, conserved, developed, managed and controlled in accordance with the requirements of the South African law (DWAF, 2004a).

To further protect the estuarine ecosystems, there has been a development of Estuary Management Plans (EMPs) with few management plans available for the estuaries in KwaZulu-

Natal, South Africa. Applicable national and provincial legislation considered in EMPs include the constitution of the Republic of South Africa (Section 24, Act 108 of 1996), National Environmental Management Act (Act 107 of 1998), 2010 Environmental Impact Assessment Regulations, Marine Living Resources Act (Act 18 of 1998) (amended 2000), National Water Act (Act 36 of 1998), National Environmental Management: Biodiversity Act (Act 10 0f 2004) (NEMBA), National Heritage Resources Act (Act 25 of 1999), Local Government: Municipal Systems Act (Act 32 of 2000), Water Services Act (No. 108 of 1997), National Environmental Management: Waste Management Act (Act 59 of 2008), Integrated Coastal Management Act (No. 24 of 2008), and National Estuarine Management Protocol.

The Integrated Water Resource Management (IWRM) is defined by the Global Water Partnership (1999) as a process which promotes and coordinates development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystem components. In addition, DWAF (2004a) defined IWRM as a process and an implementation strategy, which aims to facilitate equitable access to, and sustainable use of water resources by stakeholders at catchment, regional, national and international levels, while maintaining the characteristics and integrity of water resources at the catchment scale within established limits. The constitution of South Africa formalized these concepts in the form of the National Water Act (NWA, 1998), which details a progressive approach to water resource management in South Africa. The National Water Act (South Africa 1998) provides for sufficient water to be reserved to sustain the ecological functioning of the rivers, wetlands, groundwater and estuarine systems although the shortage of water resources has made it extremely difficult to meet this ideal (Ashton, 2007). It is thus important for water resource managers, scientists and conservationist to identify and prioritize sets of rivers or portions of rivers for conservation purposes (Knight et al., 2007).

1.6 The estuaries of KwaZulu-Natal North Coast with particular reference to uMvoti, Thukela and aMatikulu estuaries

The majority of estuarine ecosystems along the KZN north coast are threatened by the poor water quality, reduced flows and habitat alterations originating from land use activities (King & Pienaar 2011). However, the way in which estuarine ecosystems in KZN respond to these impacts is poorly understood which further limits mitigation opportunities. Estuaries in this region have been identified as significant and sensitive ecological assets (Collins and van Weele, 2013). The Thukela and uMvoti estuarine ecosystems are considered to be susceptible to external stressors from land based sources and are considered to be highly threatened with impaired ecological functions, including their ability to act as nursery grounds (Whitfield 2000; DWAF 2004b; Demetriades 2007; Turpie and Clark 2007; Van Niekerk and Turpie 2012; Collins and van Weele 2013). Increased human populations settling close to these estuaries and or upstream of these ecosystems makes these systems more sensitive to human impacts (Thomas et al. 2005; Nicolas et al. 2007; Perissinotto et al. 2010). Known anthropogenic impacts include, habitat destruction, effluent impacts, nutrient enrichment, water abstraction, overfishing as well as climate change (Harris and Kelly 1991; Allanson and Read 1995; Whitfield 2000; Collins and van Weele 2013). The aMatikulu Estuary is currently threatened from upstream impacts even though it occurs in an Ezemvelo KZN Wildlife protected area. This ecosystem has been prioritized for the development of an Estuary Management Plan (DEA 2013b) but there is no understanding of the risk from sources to ecological endpoints. The information available for the biological, ecological and physico-chemical data in the uMvoti, Thukela and aMatikulu Estuary is sparse

1.6.1 uMvoti Estuary

The uMvoti Estuary (29.40^{o}S, 31^{o}E) is located on the north coast of KZN, South Africa (Whitfield, 2000). This system is situated north of the coastal town of KwaDukuza (Stanger). The estuary is classified as a subtropical river mouth (Whitfield, 2000) with a shallow mean depth of less than 0.5 m (Begg, 1984). The uMvoti River has a mean annual run off of 15 x 10^5 m^3 (Harrison et al., 2000). This system is classified as large type-F, barred open estuary (an estuary with a supra-tidal barrier (Harrison et al., 2000). The uMvoti Estuary occupies 0.2 km^2 of the total catchment area. The uMvoti River has a catchment area of 2829 km^2 of which approximately 57 % of it is subjected to various agricultural activities (DWA, 2004). This mainly includes commercial forestry, commercial dry land agriculture (particularly sugarcane) and subsistence farming. Approximately 7 % of the catchment consists of degraded bush land and grassland while about 35 % is still natural and comprise of bushland and grassland and some forest (DWA, 2004). The uMvoti catchment land cover consists of only 1% of urban development and this is mostly residential development, smallholdings as well as industrial and commercial development associated with the town of Stanger and Greytown in the lower catchment and upper catchment respectively (DWA, 2004).

The length of the uMvoti River is 215 km with flow to the estuary ranging from 7 m^3/sec to 15 m^3/ sec (Wepener and MacKay, 2002). The uMvoti River has been categorised as a medium-sized river with a total natural mean annual runoff (MAR) of 595 million m^3/a (DWAF 2004). There is usually a 1 km long sandbar separating the estuarine area from the marine environment and it occupies approximately 20% of the total estuary area (Wepener and MacKay, 2002). Subsequently, the river deflects 90º at the coast and opens to the south. The opening of the uMvoti Estuary is over a rocky ledge of Ecca shale to the south (Begg, 1978). Since the mid-1990s, the mouth of uMvoti Estuary has seldom closed. If necessary the mouth has been breached by bulldozer to drain flooded sugar cane fields (Begg, 1978). During flooding the river takes a straighter course to the ocean and breaks through the spit in a more northerly direction (Badenhorst, 1990).

Due to a severely sedimented bed, Badenhorst (1990) calculated the mouth area of uMvoti Estuary to be 0 to 0.3 meters above sea level (m.a.s.l) which increase to 13.1 m.a.s.l in the middle reaches. Badenhorst (1990) concluded that the uMvoti Estuary has a limited potential for significant tidal exchange. Begg (1978) reported that the seawater penetration is 500 m upstream. The uMvoti Estuary has been rated severely degraded from a sedimentological point of view and this condition deteriorates with time due to high sediment loads during flooding conditions (Badenhorst, 1990).

As early as 1964, the water quality parameters of the uMvoti Estuary have been described as “grossly polluted” (Begg, 1978). Previously reported pollution was mainly due to effluent input of treated sewage from KwaDukuza via the Mbozamo Swamp, and sugar and paper mill effluents from Gledhow Sugar Mill and Sappi Stanger. Activities that are still taking place in the lower uMvoti River include treated effluent disposal, agricultural irrigation and several domestic uses by informal settlements. Sugar and paper mill operations abstract water upstream of the estuary. The uMvoti riverine system has been modified completely, with nearly total loss of natural habitat and biota and destruction of many basic ecosystem functions (Tharme, 1996). These impacts on the river are also reflected in the uMvoti Estuary. As a result, the uMvoti Estuary has been regarded as a degraded system which functions differently from the way it did in its pristine state (MacKay et al., 2000). The uMvoti Estuary has substantial recreational value.

The sand bank that separates the estuary from the ocean is normally used for launching boats. This system is a world renowned birding site and hosts some near threatened species such as *Pelecanus onocrotalus* and *Haematopus moquini* (Swemmer, 2009).

1.6.2 Thukela Estuary

The Thukela Estuary (29°13’26”S, 31°29’57”E) is a subtropical river mouth (Whitfield, 2000) located in the north coast of KZN, South Africa. The Thukela River is the second largest river in South Africa and has a catchment area of 29000 km^2 that has appropriately been named for its ferocity (Whitfield and Harrison, 2003). The estuarine area is relatively small with a surface area of approximately 0.6 km^2 with the axial length of 800 m during low flow and a shoreline length of about 2 km. The maximum width of the estuary is approximately 350 m with a channel width of 50 m (Begg, 1978).

Changes in river flows can modify the morphometrics of the Thukela estuary (Begg 1978). During floods the width of the Thukela Estuary increases to 1000 m and the estuary can extend out to the sea as there is no sea water penetration into the estuary (Begg 1978). The Thukela system is dominated by river discharge and as a result the estuary mouth is usually open (IWR Environmental, 2003). Sea water penetration was at a maximum of 1 km in occasions during 1985 and 1986. The depth of this system is 1.5 m (Archibald, 1998).

The large quantities of silt transported into the Thukela Estuary have resulted into a vertical shelf leading to minimal sea water penetration (Bosman, 2007). Sea water intrusion is most effective on spring high tide when the river flow is at minimum (from July to September) (Whitfield and Harrison, 2003). During this period, salinity of 30 can be recorded 3 km from the mouth on high tide (Whitfield and Harrison, 2003). Sediment in the mouth is composed of poorly sorted, fine, medium and coarse sand occurring at a depth of less than 30 m (Bosman et al. 2007).

Thukela River has previously experienced flood events (November 1921, March 1925, January 1934, December 1956, February 1955, September 1987) (Perry, 1989). While episodic floods have destroyed infrastructure near the Thukela Estuary in the past, these floods have been followed by improved prawn catches in the Thukela Bank fishery (Hosking, 2010). There have been changes in the upper reaches as a result of high flood conditions occurring in the Thukela River. The substrate in the upper reaches contains no cobbles or loose stones but only bedrock (Ferreira et al., 2008). Also the bedrock in this region is covered by large quantities of alien species of algae indicating anthropogenic impacts acting on this system (Ferreira et al., 2008).

The Thukela Estuary provides habitat for some marine migrants, estuarine and freshwater species and act as a conduit for many anadromic species that populate the middle and upper reaches of the Thukela Estuary (DWAF, 2004c). Another ecological importance of the Thukela Estuary includes transportation of sediment into the local marine ecosystem which is linked to the ecological functioning of a biodiversity hot-spot in the near shore marine environment and the commercially important Thukela Banks Fisheries (DWAF, 2004c). The river dominated nature of the Thukela Estuary makes it an exceptionally dynamic ecosystem which shifts between freshwater and saline states during high flow (>30 m^3/s) and low flow periods respectively, and has a rarely closed mouth state except during extremely low flow periods.

The ecological health of the Thukela Estuary and indirectly the lower Thukela River, near shore marine and Thukela Banks ecosystems has deteriorated over the last few decades. The impacts and deterioration are linked to changes in anthropogenic land use within the catchment and alteration in volume, timing and duration of flows by abstraction to meet the demand of users (DWAF 2004b). There has been an increase in pressure on the structure and function of the Thukela system as a result of increasing demand for water related ecosystem services from the Thukela River catchment (Pienaar, 2005). The lower portion of the Thukela River and the associated estuary is ecologically important region of the Thukela River catchment with several social and ecological values associated with ecosystem services (DWAF, 2004b). The state of the lower Thukela and Estuary has been reported as moderately modified (Class C) (IWR, 2004).

Anthropogenic activities associated with ecosystem services use in the lower Thukela River include water abstraction for domestic use, industries, agriculture, mining, recreation, waste water treatment and road and rail networks. Many ecosystem users abstract water directly or indirectly (via municipal abstraction works) from the Thukela and associated tributaries and some of them release treated or partially treated effluent back to the Thukela and associated tributaries e.g. Mandeni River. The region above Mandeni River outfall also supports the

Thukela-Mhlathuze Bulk Water Transfer Scheme (IWR, 2003). There is also a new Umgeni Bulk Water Transfer Scheme upstream of the Thukela Estuary. The Sappi Tugela pulp and paper mill is the major industrial activity in the area of lower Thukela and it has both the extraction and discharge points in the same region of Thukela River. Sappi releases effluent directly into the Thukela River via an underground pipe system, approximately 500m below the confluence of the Thukela and Mandeni Rivers. Reported ecological impacts as a result of the above mentioned ecosystem resource users include a drop in oxygen level along with increase in chemical oxygen demand, ammonia and conductivity e.g. (DWAF, 2004c; Ferreira et al., 2008; O'Brien and Venter, 2012; Stryftombolas, 2008).

In addition, the Mandeni River joins the Thukela River about 17 km above the Thukela River Mouth. This river is one of the ecologically important refuge areas for the aquatic animals in the lower Thukela River. In turn, the Mandeni River also receives a variety of partially treated effluents and runoff waste water from various industrial and urban centers in the region. Three of the centers in the region include the Sundumbili community, Itala (Isithebe) industrial area and Mandeni community and industrail complexes. Impacts associated with these centers include water quality impairment, water quantity alteration, habitat state alteration and disturbance to wildlife (IWR, 2003; O'Brien et al., 2009; Stryftombolas, 2008).

1.6.3 aMatikulu/Nyoni Estuary

The aMatikulu/Nyoni Estuary (36^{o}06’36”S, 31^{o}37’09”E) is a subtropical permanently open estuary located in the north coast of KZN (Whitfield, 2000). This system is in a relatively good condition although siltation from the catchment is of concern (Whitfield, 2000). The aMatikulu River joins the Nyoni River and flows parallel to the Indian Ocean before it empties into the ocean approximately 105 km north of Durban. In the lower reaches, the estuary is disturbed but the fauna remains in a relatively good condition (Whitfield, 2000). The aMatikulu/Nyoni Estuary has been reported to have a good ichthyofauna, good water quality and good aesthetics (Harrison et al., 2000). This estuary is described as a system that shares a common mouth and should be conserved as an item (Heydorn, 1985). In the surrounding area there is pioneer dune vegetation, dune forest, swamp forest, coastal riverine forest and coastal grassland with freshwater pans (Heydorn, 1985). As mentioned earlier the aMatikulu/Nyoni Estuary forms part of the nature reserve and is controlled by Ezemvelo KZN Wildlife (EKZNW).

The aMatikulu/Nyoni covers a catchment area of 900 km^2 (DWS, 2014). Agricultural activities taking place in the catchment include sugarcane and subsistence farming with some commercial forestry. Approximately 6 % of the aMatikulu/Nyoni catchment comprises degraded bushland (UDM, 2017). Approximately 60% of the catchment is under agriculture and this is predominantly sugar cane and subsistence farming and some commercial forestry (UDM, 2017). About 33 % of the catchment is natural, comprising mainly forest, grassland and bushland (UDM, 2017). Less than 1 % of the catchment is urban and this includes residential, commercial and industrial development associated with inland town of Eshowe (UDM, 2017). Threats to the aMatikulu/Nyoni Estuary are relatively minimal with few anthropogenic impacts currently. There is only one sugar mill and associated agricultural activities upstream of the aMatikulu River. Because of these reasons, the aMatikulu/Nyoni Estuary was selected as the reference site for the current study.

1.7 Regional scale risk assessment: approaches and use

1.7.1 Background

Risk assessment is defined as the process of assigning magnitudes and probabilities to the adverse effects of anthropogenic activities or natural catastrophes (Suter, 1993). These effects are referred to as hazards, and the existence of a hazard and the related uncertainty of a hazard's effects, results in the formulation of risk. Therefore, risk is the probability or likelihood of a prescribed undesired effect occurring and impacting the environment (Suter, 1993). Ecological Risk Assessment (EcoRA) can be defined as a structured approach that describes, explains and organises scientific facts, laws and relationships, thereby providing a sound basis to develop sufficient protection measures for the environment, which facilitates the development of utilisation strategies of the environment (U.S.E.P.A, 1998; U.S.E.P.A, 2008). Ecological Risk Assessment is therefore a process that evaluates the likelihood that adverse effects may occur or are occurring as a result of exposure to one or more stressors (Suter, 2001). Ecological Risk Assessment is concerned about causal relationship between stressors and effects and deals with consequences of alternative decisions (Suter, 2001).

The nature and potential of described effects of environmental stressors in terms of EcoRAs provide environmental information in a socio-economic context that drives management and environment based decision making (Suter, 2001). The EcoRA can be dated back to1983 when a so called "Red Book" was published although this book was concerned with human health and the establishment of a decision making process to assist the government of the United States of America manage risks to human health (NRC, 1983). Principles from Red Book were adopted to establish the field of non-human or environmental and ecological risks assessment from around 1987 (Landis and Thomas, 2009; NRC, 1983). The concept of carrying out regional scale risk assessment was initiated in the late 1980's and early 1990's when the risk assessment specialist group of Oak Ridge National Laboratory, Tenesse began to address ecosystem dynamics and ecological processes in risk assessments (Suter, 1993).

Environmental monitoring and EcoRAs are potentially complementary (Suter, 2001), where the data required to implement an EcoRA are provided by environmental monitoring procedures. Monitoring practises may also provide insight about the accuracy of the past EcoRAs and the effectiveness of the risk-based environmental management (Suter, 2001). Ecological Risk Assessment has been conducted successfully in other parts of the world e.g. Netherlands (Van Straalen, 2002), Japan (Tanaka, 2003) and Canada (Sadiq et al., 2003) as oposed to South Africa. In South Africa EcoRA guidelines were developed based on South African conditions and ecological stipulation information required by regional and national mandates (Claassen et al., 2001).

1.7.2 The regional risk assessment

A regional-scale ecological risk assessment can be defined as a summary of the complex interactions and effects of chemical and non-chemical stressors on the ecological endpoints (Wiegers et al., 1998). The regional risk methodology is the expansion of traditional risk assessment methodology which allows for the consideration of multiple stressors, historical events, spatial structures, and multiple endpoints in the assessment (Landis, 2005). Analyses of diverse scales, a variety of land forms and landscape types, and the integration of a diverse group of distinctly different stresses are necessary in the regional-scale methodology (Suter,1993). Furthermore, spatial and temporal variability are critical to this type of methodology. It is acknowledged that stressors occur at different locations that affect interactions and outcomes. It is also acknowledged that organisms have migration patterns as a result of life stages and seasons which alters their exposure to stressors (Wiegers et al., 1998). Indirect effects are crucial and are determined by spatio-temporal distributions of the interacting species (Wiegers et al., 1998).

There is variability in sources of stressors with season and with changes in human activity and agricultural and industrial processes (Wiegers et al., 1998). Contrary to the regional-scale ecological risk assessment, the traditional EcoRA methods only evaluate the interactions between three environmental components including stressors being released to the environment, receptors living in and using the environment and the receptor's response to the stressors that constitute a risk (Fig. 1.1a) (Wiegers et al., 1998). Measurements of exposures and effects allows for quantification of the degree of interaction between these components. In one contaminated site where there is usually only one stressor involved, the connection of the exposure and effect measurements to the endpoints may be relatively simple (Wiegers et al., 1998). However, in a regional multiple-stressor assessment there is an increase in the number of possible interactions.

Stressors originate from diverse sources, receptors are usually associated with a variety of habitats and eventually one or more impacts may result (Figure 1.1b) (Wiegers et al., 1998). In regional scale risk assessment, complexity of structure and the regional spatial components of sources that release stressors, habitats where receptors reside and impacts to the assessment endpoints are required (Fig. 1.1b). At a regional level sources and stressors can be represented as groups. A habitat as a group of receptors and an ecological impact as a group of receptor responses. All these spatial components allow for the consideration of risk at a complex scale (Landis and Wiegers, 1997; Wiegers et al., 1998). The three spatial components (sources, habitats and impacts) are similar to the traditional components (stressors, receptors and responses) but emphasis is on the locations and allows for the consideration of groups of stressors, receptors and responses or effects (Landis, 2005). The combination of stressors, receptors and responses which makes up the groups within regional EcoRA can be made up of a variety of different measurements that can be difficult to compare. To solve this problem, the regional EcoRA methodology makes use of relative ranks to combine these measurements (Landis, 2005). Relative regional assessment method allows for the identification of sources and habitats in different regions of the site, ranks their importance in each location and combines this information to predict relative levels of risk (Wiegers et al., 1998).

1.7.3 The Relative Risk Model

A Regional Scale Risk Assessment using the Relative Risk Model (RRM) is a form of EcoRA that is carried out on a spatial scale where considerations of multiple sources of multiple stressors affecting multiple endpoints are allowed (Landis, 2005). In addition, allowance for the landscape characteristics that may affect the risk estimate is made possible. Similar to traditional EcoRA approaches, RRM may be carried out to assess the risk posed by one stressor of concern to one endpoint. However, at a regional scale, multiple stressors acting on a range of ecological endpoints can be considered within a RRM framework (Landis and Wiegers, 1997).

The number of possible risk combinations depends on the number of categories identified for each component. For example, if two source types and two habitat types are identified, then four possible combinations of identified components may have an impact. If two different impacts are of concern then eight combinations may result in a potential environmental risk (Fig. 1.2) (Wiegers et al., 1998). For every identified combination, there is establishment of a possible pathway to a risk in the environment. This method thus allows for the evaluation of multiple stressors being derived from multiple sources and impacting on a variety of species in a variety of habitats and a variety of locations (Landis, 2005). If a certain combination of components affects each other/ interact then they can be thought of as overlapping. When a source produces stressors that affects the habitats utilised by and important to the components of endpoints, the ecological risk is high (Fig. 1.3a). A minimal risk is obtained when there is a slight interaction between components (Fig. 1.3b). If one component does not interact with one or the other two components, then no risk will be posed to the environment (Fig. 1.3c & d). Figure 1.2 depicts that impact 1 appears in four combinations and that each combination can overlap. When these combinations are integrated, it is evident that impact 1 is actually the result of several combinations of sources and habitats (Fig. 1.4). For better understanding of a risk of a single impact occurring, each possible route to the impact needs to be investigated (Wiegers et al., 1998). However, it is not always easy and simple to integrate these routes (Wiegers et al., 1998). It is always difficult to add together the measurements of various exposure and effect levels to determine the overall impact to assessment endpoints because of different metrics used to quantify the various impacts.

The regional risk assessment using relative risk model involves a system of numerical ranks and weighing factors to address the difficulties encountered when attempting to combine different kinds of risks (Fig. 1.5) (Landis, 2005). Ranks and weighing factors are unitless measures that operate under different limitations when compared with measurements with units (e.g. mg/l, individuals/cm^2) (Wiegers et al., 1998). Scale of 1 and 0 is used to weigh combinations resulting in exposures and effects to indicate overlap and interaction among the stressors, habitats and effects (Fig. 1.5 b & c) (Landis, 2005). The RRM is known to be a useful approach that can contribute towards management of surface aquatic ecosystems in South Africa for the protection of biodiversity while allowing for the social and economic needs of the society (O'Brien, 2012).

1.7.4 The ten steps of the Relative Risk Model for Regional Risk Assessment

To carry out regional ecological risk assessment, the following ten steps need to be performed (Landis, 2005).

1. Vision exercise: List the important management goals for the region. What do you care about and where?
2. Mapping and data analyses: Include potential sources and habitats relevant to the established management goals.
3. Risk region selection: Section map into regions based on the combination of the management goals, sources and habitats.
4. Conceptual model: Construct a conceptual model that links the sources of stressors to receptors and to the assessment endpoints.
5. Ranking scheme: Decide on ranking scheme to allow the calculations of relative risk to the assessment endpoints.
6. Calculate risks: Calculate the relative risks.
7. Uncertainty evaluation: Evaluate uncertainty and sensitivity analysis of the relative rankings.
8. Hypotheses establishment: Generate testable hypothesis for future field and laboratory investigation to reduce uncertainties and to confirm risk rankings.
9. Test hypotheses: Test hypotheses in step 8
10. Communicate outcomes: Communicate the results in a fashion that portrays the relative risk and uncertainty in response to the management goals.

The first four steps are critical to performing the EcoRA and they serve as the foundation of the assessment that will later allow for the outcomes of the assessment to be used in the decision making process and long term environmental management. In these steps, there should be a close interaction with all of the stakeholders of an assessment. The final step (step 10) deals with communicating the outcomes of the study and this can take three forms (Landis, 2005):

1. Maps of the risk regions with associated sources, land uses, habitats and the spatial distribution of the assessment endpoints.
2. A regional comparison of relative risks, their causes, patterns of impact to assessment endpoints and the associated uncertainty.
3. A source-habitat-impact model that can be used to produce scenarios pertaining the environmental management options for the study area.

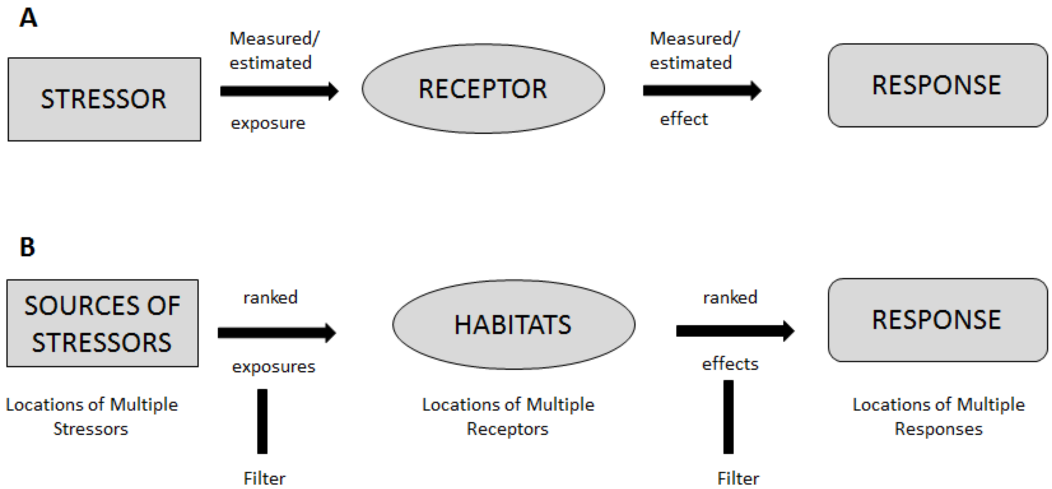


Fig 1.1: Comparison of traditional risk assessment (A) with regional scale relative risk assessment (B) (Adapted from Landis, 2005).

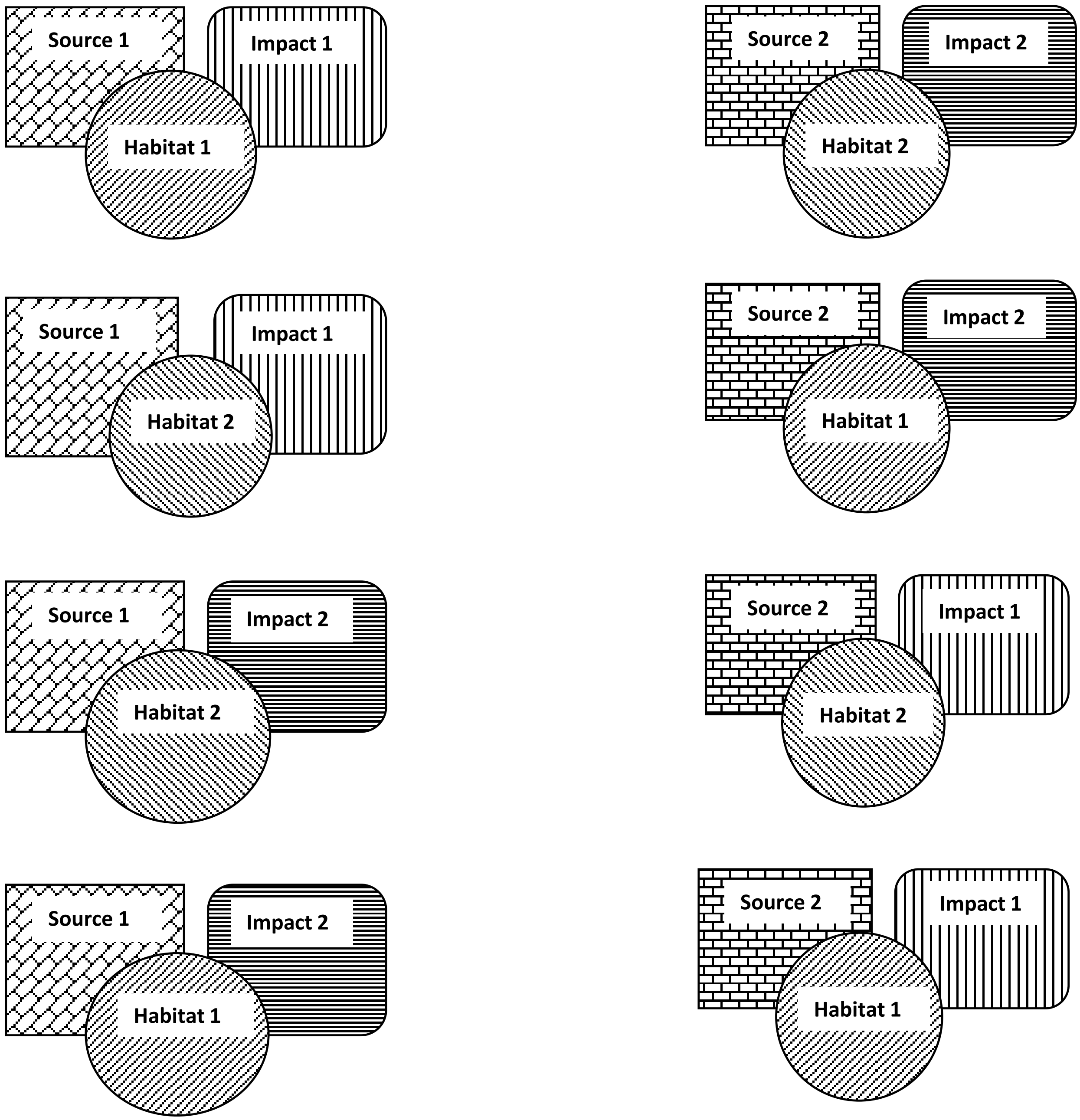


Fig 1.2: Possible combinations characterising risk from two sources, two habitat types and two potential impacts to assessment endpoints. Potential impacts are those that affect the assessment endpoints that may occupy the specific habitat (Adapted from Landis and Wiegers, 1997).

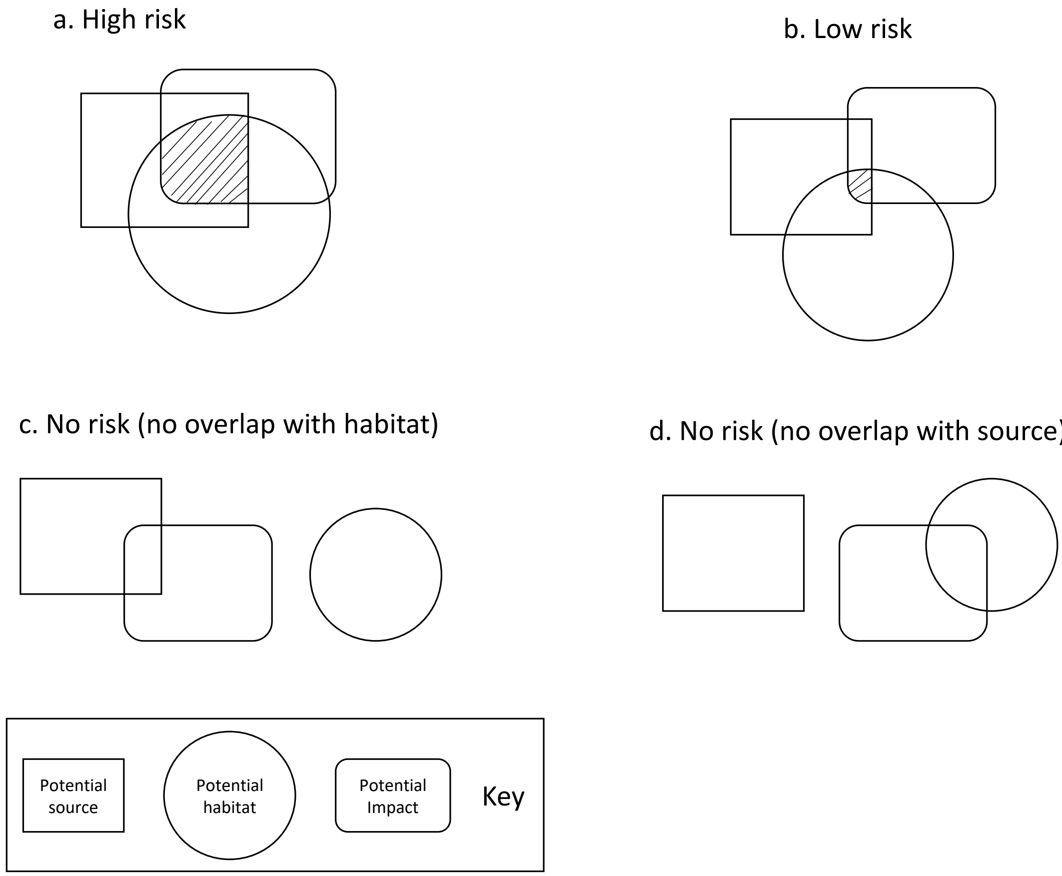


Fig 1.3: Ecological risk resulting from the interaction between sources, habitats and endpoints in the environment. Risk is assumed to be proportional to the overlap of source, habitat and impact (Adapted from Landis and Wiegers, 1997).

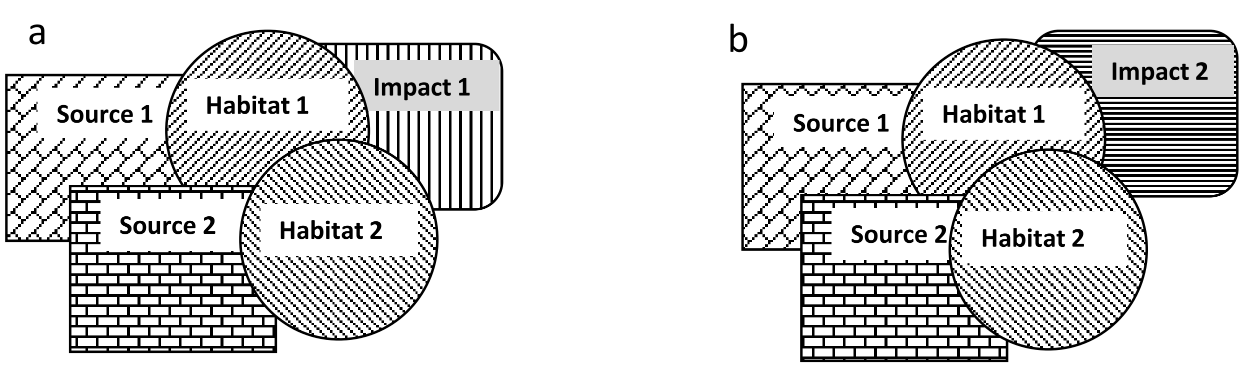
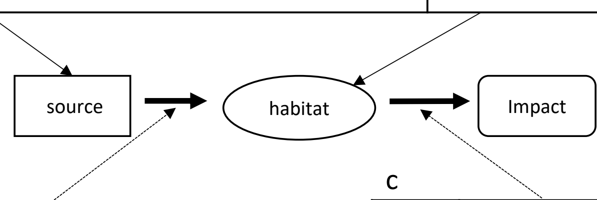


Fig 1.4: Different interactions through overlap of the possible combinations of two sources and two habitat types that can influence the risk posed to assessment endpoints presented as impact 1 (a) and impact 2 (b) (Adapted from Landis and Wiegers, 1997).

a

Ranks	Source type	Habitat type
6	High discharge or activity from the source in the sub-area	Large amount of the habitat in the sub-area
4	Moderate discharge or activity from the source in the sub-area	Moderate amount of the habitat type in the subarea
2	Low discharge or activity from the source in the sub-area	Small amount of the habitat type in the sub-area
0	No sources of this type in the area	No habitats of this type in the area



b

Score	Exposure combination
0	The source is unlikely to occur or be transported into the habitat source habitat
1	The source is likely to occur or be transported into the habitat source habitat

c

Score	Effect combination
0	The impact is unlikely to occur in the habitat or because of the source source habitat Impact
1	The impact is likely to occur in the habitat or because of the source source habitat Impact

Fig 1.5: The application of ranks and filters in the Relative Risk Model framework (Adapted from Landis, 2005).

1.8 Problem statement and significance of the study

The continued deterioration in the ecological state of South Africa's surface aquatic ecosystems is causing an inevitable decline in the provision of key ecosystem services upon which the social and economic development of the country depends (Ashton, 2007; Driver et al., 2005). At present only approximately 30% of South Africa's main rivers are still in good ecological state and sustainable while a surprising 47% have been modified to varying degrees and 23% have been irreversibly transformed (Nel et al., 2005). Estuaries are among the most productive and dynamic ecosystems globally, and have a high ecological, economic and social value (Chuwen et al., 2009b; McLusky, 2004; Vasconcelos et al., 2010). The ecological value of these systems includes the unique processes and the provision of nursery grounds for many marine species

(Beck et al., 2003; Elliott and McLusky, 2002; Nicolas et al., 2007; Potter and Hyndes, 1999; Vasconcelos et al., 2010). The estuarine environment benefits these organisms through the provision of suitable conditions needed for growth such as sheltered habitat, suitable water temperature and high food availability (Wallace and van der Elst, 1975; Miller et al., 1985; Beck et al., 2001). Such an estuary nursery phase in marine species is compulsory for many marine species in order for them to complete their life cycle and maintain marine populations (Perillo, 1995; Strydom and Whitfield, 2000; Vasconcelos et al., 2010; Whitfield, 1999a). Estuarine systems serve as filters because when there is high level of nutrients entering the estuary from land drainage the estuary traps the extra nutrients that are not utilized by biota, stores them in sediment and make these available in water column later when they are needed by biota through recycling (Scharler and Baird, 2005; Taljaard et al., 2009; Telesh and Khlebovich, 2010). Social benefits include available proteins to humans as they fish in these estuaries for food and obtain recreational benefits.

The structure, function, processes and biodiversity of estuarine ecosystems are vulnerable to threats from local and catchment scale land use activities (O'Brien et al., 2009; O'Brien and Wepener, 2012; Stewart-Koster et al., 2010). The understanding of how threats (stressors) associated with multiple land use activities (sources of stressors) impact on the ecological function and processes of ecosystems in KZN in relation to conservation and management requirements (endpoints) is urgently needed. This must include considerations of estuarine ecosystem dynamics, be transparent, adaptable and make use of international best scientific practice methods. In South Africa, 79% of the estuarine area is in a threatened state (DEA, 2013a). Only 1% of the total estuarine habitat area in South Africa is classified to be in an excellent condition with 14% in a good condition and 31% in a fair condition (DEA, 2013a). Fifty four percent of estuaries in South Africa are in poor condition. At present, about 59% of South African estuaries are not protected (Van Niekerk and Turpie, 2012). Although the ecosystem services provided by estuaries in KZN are poorly documented (Andersson et al., 2009; DEA, 2013a; DWAF, 2004b; O'Brien and Venter, 2012; O'Brien, 2012) these estuarine systems are known to provide both the environment and people with multiple benefits. These benefits include natural products (water, fish, sand and vegetation), land derived nutrients, sediment and food resources to the near shore marine environment, conduit for the movement of marine and freshwater species between these ecosystems and waste assimilation functions (de

Groot, 1994; Diaz et al., 2004; DWAF, 2004c; Gillanders and Kingsford, 2002; Peterson and Ross, 1991; Pittman and McAlpine, 2001; Vasconcelos et al., 2010; Wasserman and Strydom, 2011).

The majority of estuarine ecosystems along the KZN north coast are threatened by the poor water quality, reduced flows and habitat alterations originating from land use activities. The way in which estuarine ecosystems in KZN respond to these impacts however are poorly understood which limits mitigation opportunities. Local conservationists and managers need to understand the relative importance of different ecosystem processes, the relative contribution of threats from multiple sources to receptors in relation to management endpoints. Information gained from the current study will allow for impacts to be addressed through proposing management options and targets based on the sources of impacts and by proposing effective mitigation measures. Estuaries in the north coast of KZN have been identified as significant and as sensitive ecological assets (Collins and van Weele, 2013). The information from the current study will contribute to the legal requirements of municipal, district and national regulators to establish a suitable balance between the use and protection of these ecosystems (DEA, 2013a; DEA, 2013b).

The Thukela and uMvoti estuarine ecosystems are considered to be susceptible to external stressors from land based sources and are considered to be among the more highly threatened estuaries with impaired ecological functions, including their ability to act as nursery grounds (Collins and van Weele, 2013; Demetriades, 2007; DWAF, 2004b; Turpie and Clark, 2007; Van Niekerk and Turpie, 2012; Whitfield, 2000). These systems are getting more vulnerable to anthropogenic impacts as human population settling in their catchments and industries are continuously increasing (Nicolas et al., 2007; Perissinotto et al., 2010; Thomas et al., 2005). These impacts may include water abstraction, nutrient enrichment, habitat destruction, overfishing and effluent discharges (Allanson and Read, 1995; Collins and van Weele, 2013; Harris and Kelly, 1991; Whitfield, 2000). The aMatikulu Estuary is currently threatened from upstream impacts even though occurs in an Ezemvelo KZN protected area. This ecosystem has been prioritized for the development of an Estuary Management Plan (DEA, 2013b) but there is no knowledge of the risk from sources to ecological endpoints. An outdated Ecological Reserve and a draft Estuary Management Plan has been established for the Thukela Estuary (DWAF 2004a; DEA 2013b), but the risk of freshwater deprivation and other upstream impacts are poorly understood and needed to manage this ecosystem effectively. The information from this study will make a considerable contribution to the management of the balance between use and protection of estuaries in KZN which is currently skewed towards use. If the balance between the use and protection of these estuaries is not established, some ecosystem services, ecological processes and biodiversity in these systems will be lost. In addition, habitat transformation and loss may result. If estuarine resources are highly utilised or exploited, availability of services will be threatened with no equitable access to the resources and these systems will continue to deteriorate. The big problem in management of estuaries is the conflict between use (e.g. water abstraction, waste discharge) and the protection of the natural environment. The increased use of water resources upstream will result in a negative alteration of river flow which influences marine fish and fisheries through the export of nutrients, sediment and detritus (Baird and Heymans, 1996; Gillanders and Kingsford, 2002; Loneragan, 1999; Robins et al., 2005). Therefore, evaluating the risk which the developmental services and other anthropogenic impacts pose into estuarine systems is essential to understand the extent of impacts in these systems and to develop the appropriate management strategies. The present study seeks to answer the following question: What are the risks (and the extent) associated with land use activities in the uMvoti, Thukela and aMatikulu estuaries and which ecological endpoints are highly threatened to qualify for management/conservational priority.

1.9 Aims and objectives

The following aim was established to address the research question posed:

To carry out a Regional Scale Risk assessment incorporating Bayesian Network probability modelling techniques to evaluate the threat of upstream (catchment) and local (iLembe district municipal boundary) land use activities as well as threat of flow alterations to the ecological and selected social objectives/endpoints of the uMvoti, Thukela and aMatikulu estuaries.

To achieve the aim of the study, the following objectives have been established:

- Quantify and compare the composition (spatial and temporal trends) of macrozoobenthos communities within and between the uMvoti, Thukela and aMatikulu estuaries using multivariate statistical analyses. The hypothesis tested is that the macrozoobenthos community within and between the three estuaries would vary along the spatio-temporal scale, and that the differences would be related to variations in catchment land use patterns and water quality.

- Quantify and compare the composition (spatial and temporal trends) of zooplankton communities within and between the uMvoti, Thukela and aMatikulu estuaries using multivariate statistical analyses. The hypothesis that the zooplankton community within and between the three estuaries would vary along the spatio-temporal scale, and that the differences would be related to variations in catchment land use patterns and water quality was tested.
- Carry out a Regional Scale Risk Assessment for the uMvoti, Thukela and aMatikulu estuaries to evaluate the threat of land use activities to selected protection endpoints for the study area. The hypothesis that the protection endpoints will display different risk scores between the three estuaries as a result of different anthropogenic land use activities posing different stress levels in these systems was tested.

1.10 Study outline

The thesis consists of five chapters of which three data chapters are prepared for submission to relevant international peer-reviewed journals, and thus some repetition in other chapters was unavoidable. The arrangement of the chapters is as follows:

Chapter 2: Macro-benthic invertebrate communities in selected river dominated estuaries in KwaZulu-Natal, South Africa: effects of altered water quality and quantity

Chapter 3: Effects of land use activities on zooplankton of selected river dominated estuaries in KwaZulu-Natal, South Africa.

Chapter 4: Application of Relative Risk Model for evaluation of ecological risk in selected river dominated estuaries in KwaZulu-Natal, South Africa.

Chapter 5: The concluding chapter that summarizes the different components of the study and provides conservation and management recommendations for the KwaZulu-Natal estuaries.

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CHAPTER 2

Macro-benthic invertebrate communities in selected river dominated estuaries in KwaZulu-Natal, South Africa: effects of altered water quality and quantity

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Abstract

Globally estuaries are important, but many are threatened by anthropogenic activities. We quantified and compared the composition of macrozoobenthos communities within and between three estuaries (uMvoti, Thukela and aMatikulu/Nyoni estuaries) in KwaZulu-Natal, South Africa, with different anthropogenic land use. We also related the macrozoobenthos of each to their respective physico-chemical environments. The subtidal macrozoobenthos of these estuaries were sampled in 2014 - 2015. The Thukela Estuary had the highest number of macrozoobenthos taxa (24) followed by aMatikulu/Nyoni (11) and uMvoti (8). In contrast, macrozoobenthos abundance was highest in the aMatikulu/Nyoni Estuary followed by Thukela and then uMvoti Estuary. The Thukela and aMatikulu/Nyoni estuaries contained freshwater, estuarine and marine taxa, while the macrozoobenthos of the uMvoti Estuary was mostly dominated by freshwater taxa. Macrozoobenthos abundance decreased from the upper to the lower reaches in all three estuaries. Redundancy analysis (RDA) plots revealed that higher water temperature, turbidity and phosphate levels affected the structuring of the macrozoobenthos community in the uMvoti and Thukela estuaries, while in the aMatikulu/Nyoni Estuary higher salinity, pH and oxygen were responsible for the structuring of the macrozoobenthos community. Although the estuaries are from the same geographical region with similar form and function, the high variability in their macrozoobenthos communities was explained by differing anthropogenic land use activities in their catchments.

Keywords: Community structure, estuarine ecological state, ecological indicators, anthropogenic land use activities, macrozoobenthos, water quality

2.1 Introduction

Estuaries are among the most productive and dynamic ecosystems globally, and have high ecological, economic and social value (McLusky 2004; Chuwen et al. 2009; Vasconcelos et al. 2010). The ecological value of this ecotone interface of freshwater and marine systems includes the unique processes, species diversities present, and the provision of nursery grounds for many marine species (Vasconcelos et al., 2010). The distribution of fauna in estuaries has been reported to be controlled by salinity primarily, and secondarily by substrate, temperature, dissolved oxygen and anthropogenic pollution (Kinne 1996; Harrison and Whitfield 2006). Hypoxia, related to circulation of bottom waters is also known to affect estuarine macrozoobenthos by changing community composition and reducing biodiversity and biomass (Holland et al. 1987; Rabalais and Harper 1991; Dauer et al. 1992; Teske and Wooldridge 2003). In organically polluted estuaries, macrozoobenthos is mostly dominated by small deposit feeders which are characteristic of polluted waters (Herman et al. 1999).

Globally there has been a serious deterioration in the ecological health of many estuaries as a result of anthropogenic activities including excessive water abstraction, agricultural and industrial effluents (Owens 1991; Kennish 2002; Quinton and Catt 2007; Zhang et al. 2012). Estuarine ecosystems are dependent on riverine freshwater input of sufficient quantity (Kimmerer 2002). However, increasingly estuaries receive major anthropogenic input generated further upstream from point and non-point sources, and from urban areas and industries near these systems (Chapman and Wang 2000). Fifty-four percent of estuaries in South Africa are in relatively poor condition (Van Niekerk and Turpie 2012). Furthermore, at present most (~59%) of South African estuaries are not protected (Van Niekerk and Turpie 2012). The majority of estuarine ecosystems in the north coast of KwaZulu-Natal (KZN) Province are threatened by the poor water quality, reduced flows and habitat alterations originating from land use activities (King and Pienaar 2011). The ways in which estuarine ecosystems in KZN respond to these impacts, however, are poorly understood which limits mitigation and conservation opportunities. The understanding of how threats (stressors) associated with multiple anthropogenic land use activities (sources of stressors) impact on the ecological structure, function and processes of estuarine ecosystems in KZN in relation to conservation and management requirements (endpoints) is urgently needed.

The structure, function, processes and biodiversity of estuarine ecosystems are threatened by the local and catchment scale anthropogenic land use activities (Stewart-Koster et al. 2010; O'Brien and Wepener 2012). The uMvoti Estuary, KZN, has been rated severely degraded in terms of sedimentology and this condition deteriorates with time due to high sediment loads during flooding conditions (Badenhorst 1990). The ecological functioning of the Thukela Estuary, KZN, and indirectly the lower Thukela River, near shore marine and Thukela Banks ecosystems have deteriorated over the last few decades (DWAF 2004). The impacts and deterioration are linked to changes in land use within the catchment and alteration in volume, timing and duration of flows by abstraction to meet the demand of users (DWAF 2004). In contrast the aMatikulu/Nyoni Estuary is considered to be in relatively good condition although siltation from the catchment is of concern (Whitfield 2000). As a consequence, the aMatikulu/Nyoni Estuary was selected as the reference site for the present study.

Macrozoobenthos communities have been identified as principal components in the functioning of estuarine ecosystems because of their high contribution and importance in the structuring of estuarine food webs (Herman et al. 1999; Gray and Elliot 2010). These macrozoobenthos communities support higher trophic levels, including crabs, prawns, fish, and birds, in many estuaries and adjacent marine environments (Barry et al. 1996). Macrozoobenthos are the main food source for many estuarine fish species (Gibson 1994). This is also one of the principal forces driving tidal migrations of fish in estuarine waters (Gibson 1994; Vinagre et al. 2006). Furthermore, macrozoobenthos organisms have been identified as suitable ecological indicators in estuaries detecting the effects of stress and pollution (Stark et al. 2003; Salas et al. 2006; Patricio et al. 2009; Pinto et al. 2009) as well as water and sediment quality (Chapman and Wang 2000; Dauer and Ranasinghe 2000; Sarang and Sharma 2009).

In this study we quantified and compared the composition of macrozoobenthos communities within and between the three estuaries (uMvoti, Thukela and aMatikulu/Nyoni estuaries) in KZN with different catchment anthropogenic land use activities. We also related the macrozoobenthos community structure of each estuary to their respective physico-chemical environments. As the three estuaries are similar in form, and geographical area, we hypothesized that the macrozoobenthos community within and between the three estuaries would vary along the spatio-temporal scale, and that the differences would be related to variations in catchment anthropogenic land use, and water quality.

2.2 Methods

2.2.1 Study sites

Three estuaries (uMvoti, Thukela and aMatikulu/Nyoni) along the KZN north coast, South Africa (Fig. 2.1), were selected for this study. The three estuaries considered in this study are river dominated and the uMvoti and Thukela estuaries are considered as river mouths while the aMatikulu/Nyoni Estuary is considered as a permanently open estuary (Whitfield 2000).

2.2.1.1 uMvoti Estuary

The uMvoti Estuary (29^{o}23’ S, 31^{o}20’ E) is situated north of the coastal town of KwaDukuza (Stanger) (Fig. 2.1). The estuary is classified as a subtropical river mouth (Whitfield 2000). The uMvoti Estuary occupies 0.2 km^2 of the total catchment area. The uMvoti River catchment is subject to agricultural activities which include commercial forestry, sugar cane farming, commercial dry land agriculture and subsistence farming (pers. obs.). There is usually a 1 km long sandbar separating the estuarine area from the marine environment and it occupies approximately 20% of the total estuary area (Wepener and MacKay 2002). The uMvoti Estuary has a limited potential for significant tidal exchange (Badenhorst 1990). The seawater penetration is 500 m upstream (Begg 1978). The uMvoti Estuary has a substantial recreational value (O’Brien et al. 2009).

2.2.1.2 Thukela Estuary

The Thukela Estuary (29^{o}13’ S, 31^{o}29’ E, Fig. 2.1) is a subtropical river mouth (Whitfield 2000). The Thukela River is the second largest river in South Africa and has a catchment area of 29000 km^2 that has appropriately been named for its ferocity (Whitfield and Harrison 2003). The estuarine area is relatively small with a surface area of approximately 0.6 km^2. Begg (1978) reported that changes in river flows can modify the morphometrics of this estuary. During floods, the Thukela Estuary extends out into the sea as no sea water can penetrate the estuary (Begg 1978). The dominant physical process in the estuary is the river discharge and as a result the estuary mouth is usually open (IWR Environmental 2003). The large quantities of silt transported into the Thukela Estuary have resulted into a vertical shelf leading to minimal sea water penetration (Whitfield and Harrison 2003; Bosman et al. 2007). Sea water intrusion is highest during spring high tide especially when the river flow is at minimum (from July to September) (Whitfield and Harrison 2003). During this period, salinity of 30 can be recorded 3 km upstream from the mouth during high tides (Whitfield and Harrison 2003).

The Thukela Estuary provides habitat for some marine migrants, estuarine and freshwater species, and acts as a conduit for many anadromous species that populate the middle and upper reaches of the Thukela Estuary (DWAF 2004). Another ecological importance of the Thukela Estuary is the transportation of sediment into the local marine ecosystem which is linked to the ecological functioning of a biodiversity hot-spot in a near shore marine environment and the commercially important Thukela Bank Fisheries (DWAF 2004). The river dominated nature of the Thukela Estuary makes it a dynamic ecosystem which shifts between freshwater and saline states during high flow (>30 $m^3 \cdot s^{-1}$) and low flow periods respectively, and it is rarely in a closed mouth state except during extremely low flow periods (DWAF 2004).

2.2.1.3 aMatikulu/Nyoni Estuary

The aMatikulu/Nyoni Estuary (36°06’ S, 31°37’ E, Fig. 2.1) is a subtropical permanently open estuary (Whitfield 2000). The aMatikulu River joins the Nyoni River and flows parallel to the Indian Ocean before it empties into this ocean approximately 105 km north of Durban. The aMatikulu/Nyoni Estuary covers a catchment area of 900 km^2 (Begg 1978). In the lower reaches, the estuary is disturbed by agricultural activities but the fauna generally remains in a good condition (Harrison et al. 2000). The aMatikulu/Nyoni Estuary has relatively good ichthyofauna, good water quality and good aesthetics (Harrison et al. 2000). This system is described as a system that shares a common mouth and should be conserved as an item or as one system (Heydorn 1986). Furthermore, as the surrounding area has pioneer dune vegetation, dune forest, swamp forest, coastal riverine forest and coastal grassland with freshwater pans, it is of conservation importance (Heydorn 1986). The aMatikulu/Nyoni Estuary forms part of the nature reserve, managed by Ezemvelo KwaZulu-Natal Wildlife (EKZNW).

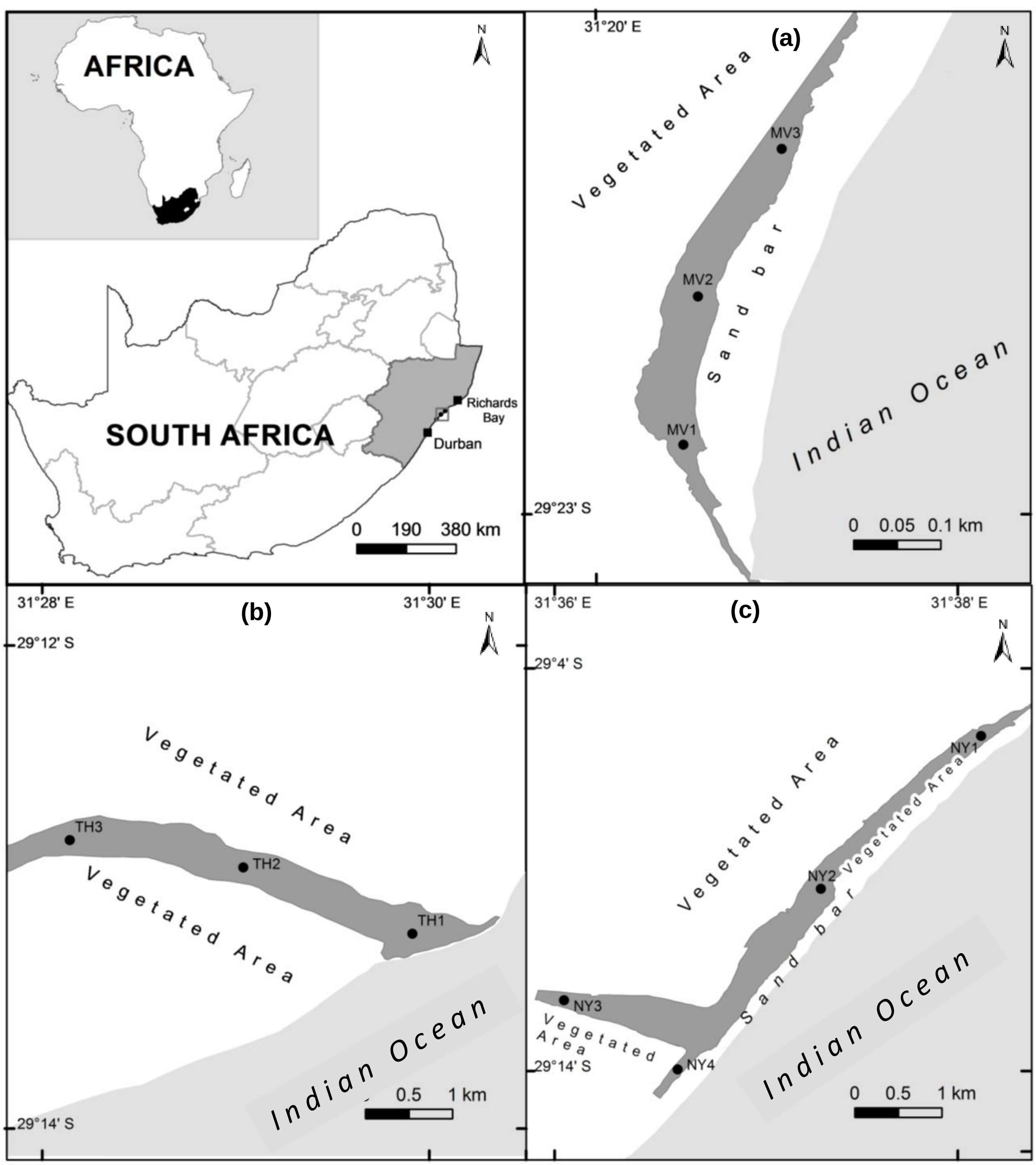

Fig. 2.1: The uMvoti (a), Thukela (b) and aMatikulu/Nyoni (c) estuaries with sampling sites occupied at each system during the study period. MV1-3 = uMvoti Estuary site 1-3; TH1-3 = Thukela Estuary sites 1-3, NY1-4 = aMatikulu/Nyoni Estuary sites 1-4.

2.2.2 Sampling and laboratory analysis

Macrozoobenthos were sampled in the uMvoti, Thukela and aMatikulu/Nyoni estuaries during 2014 (August) and 2015 (March and August). Three sites were sampled in uMvoti and Thukela estuaries and four sites in aMatikulu/Nyoni Estuary (Fig. 2.1). For temporal comparisons, previously available macrozoobenthos data collected in 2005 March and August in the uMvoti and aMatikulu/Nyoni estuaries and 2013 September in the Thukela Estuary were also included in the current study. No sampling was performed in aMatikulu/Nyoni Estuary during 2014.

During each survey *in situ* water quality data including oxygen, pH, salinity, temperature, conductivity and total dissolved solids (TDS) were recorded using a portable water meter (Eutech instruments CyberScan series 600, Thermo Fisher, USA). Water samples for further water quality analyses were collected from the subsurface using polyethylene bottles and were sent to Umgeni Water Laboratory (an accredited laboratory with the South African National Accreditation System and the International Standard ISO/IEC 17025:2005) for analysis. Three replicate samples, comprising five grabs each, were collected using a Zabalocki-type Eckman grab which samples a uniform area of 0.024 m^2 to a maximum depth of 15.5 cm. Replicate sediment samples were emptied into 25 l buckets and were stirred vigorously to suspend benthic organisms and supernatant decanted through a 500 microns sieve. Prior to the stir and decant process ~15 ml of 30% formalin was added to each bucket to coerce the organisms to release the sediment. The process of stirring and decanting was repeated five times and this procedure has been reported to extract 95% of organisms in samples (Cyrus and Martin 1988). All retained organisms in each sample were then stored in 250 ml plastic honey jars and preserved in 10% formalin containing Rose Bengal dye to aid sorting in the laboratory. We checked the remaining sediment for larger and heavy organisms, such as molluscs, which were then added to the respective sub-samples. During March and August 2015 sampling, the high density of reeds and grass in the middle reaches of aMatikulu/Nyoni Estuary (NY2 and NY3) prevented boat access. As a result, data from these two sites in the aMatikulu/Nyoni Estuary during March and August 2015 are absent.

In the laboratory, all macrozoobenthos organisms collected during surveys were sorted and identified to the lowest taxa possible and counted. Most groups were identified to family level since it is acknowledged that identification to family level provides sufficient taxonomic resolution for detecting environmental change in strong environmental gradients (Warwick

1988a; b). Taxa were identified under a dissecting microscope using Day (1967b, 1974b), Griffiths (1976), Kensley (1978) and Steyn and Lussi (1998) as guides. At each sampling site, final abundance was expressed as mean number of each taxon per m^{-2} of substratum. Sediment grain size composition was determined using classical dry sieving method, which separates sediment grains according to their intermediate axial length (dI) (Blair & McPherson 1999). Samples were dried at 60 °C in a Labcon oven for 24 h. Total dry mass of the sediment was measured with an electrical micro scale (accurate to two decimal places). Sediment was then run through a sieve stack of decreasing mesh sizes (between 2000 and 63 µm) and mechanically shaken for a standard duration of 10 min. Mass of sediment retained by each sieve was weighted and represented as a proportion of the total sediment. Dry sieving method was used to get the cumulative percentage weights of gravel (>2 mm), very course sand (2–1 mm), course sand (1–0.5 mm), medium sand (0.5–0.25 mm), fine sand (0.25–0.125 mm), very fine sand (0.125–0.063 mm) and silt (<0.063 mm) sediment fractions. The percent organic content was determined by oven drying a sediment sample of approximately 5 g which was then incinerated at 600 ºC for 6 h (Gray 1981).

2.2.3 Statistical analysis

In the current study we applied ordination techniques using the original macroinvertebrate community data sets for analysis (Van den Brink et al. 2003). This allowed for the direct interpretation of the community structures of macroinvertebrates in terms of the taxa obtained in the current study. These techniques allowed assessment of complex responses or changes in community structures and then when combined with Monte Carlo permutation testing, the statistical significance of hypothesised differences in the community structures could be tested (Van den Brink et al. 2003; Ter Braak and Smillauer 2004). Initially the ordination approach allowed for the expression of macroinvertebrate taxa between sites and surveys without the need for correlating environmental or explanatory data. This allowed for the spatial and temporal variations of the composition of macroinvertebrate taxa to be considered in an optimised form to reflect the underlying structure of the data set (Ter Braak 1994). In this ordination approach, the largest part of the total variance of the data sets were used to establish a first latent variable and then a second which relied on the largest part of the remaining variance in the data set (Van den Brink et al. 2003). We used these two latent variables to construct an ordination diagram forming two axes. Samples (sites per survey) and taxa were initially presented in the diagram as points at the location of the values of the latent variables. Samples with nearly identical or similar taxa compositions were located close together while samples located far apart represented those samples that had differing compositions of taxa (Van den Brink et al. 2003). Thereafter by including available explanatory environmental data, tri-plots were established that presented arrows of environmental data which pointed in the direction of higher values where correlations between the environmental variables and the sites occurred (Van den Brink et al. 2003). In this study we used constrained analyses, which involved overlaying the captured variance of the explanatory environmental variables onto macroinvertebrate and taxa ordination diagrams. The linear response used to achieve this was a redundancy analysis (RDA), a derivative of principle component analyses (PCA) using the CANOCO version 4.5 software package (Ter Braak 1994). Because abundance data were available, the data were transformed using a Log X+2 - transformation (Van den Brink et al. 2003). The redundancy analyses were also performed to detect if there were any significant differences between sites, years and flows.

2.3 Results

Physico-chemical data including temperature, oxygen, conductivity, salinity, pH and turbidity were compared along the salinity gradient and between the sampling sessions for the uMvoti, Thukela and aMatikulu estuaries. Macrozoobenthos organisms were analysed and their number of taxa and abundance were compared along the salinity gradient, between sampling sessions and between estuaries. The three estuaries compared in the current study are affected by different anthropogenic land use activities of varying intensities in their catchments. The uMvoti Estuary is impacted by treated effluent disposal, agricultural irrigation and several domestic uses by informal settlements while the Thukela Estuary is impacted by water abstraction for domestic use, industries, agriculture, mining, recreation, waste water treatment and road and rail networks. In contrast the aMatikulu Estuary has minimal impacts with one sugar mill and associated agricultural activities upstream.

2.3.1 Environmental variables

Measured physico-chemical parameters including temperature, dissolved oxygen, conductivity, salinity, pH, and turbidity in the uMvoti, Thukela and aMatikulu estuaries during the current study are presented in Table 2.1.

In the uMvoti Estuary the lowest temperature (17 ^{o}C) was recorded during the 2005 low flow while the highest (29.6 ^{o}C) was recorded during the 2005 high flow. In Thukela Estuary the lowest temperature was recorded during 2014 (17.5 ^{o}C) in the lower reaches while the highest was recorded during 2015 (30.6 ^{o}C) in the middle reaches. In the aMatikulu Estuary the lowest temperature (19.3 ^{o}C) was recorded during 2005 in the middle reaches while the highest (29.4 ^{o}C) was recorded during 2015 in the upper reaches. In all the three estuaries studied, temperature generally increased from the lower to the upper reaches.

In the uMvoti Estuary the lowest oxygen content (1.9 $mg \cdot l^{-1}$) was recorded during 2005 low flow in the upper reaches while the highest (6.5 $mg \cdot l^{-1}$) was recorded during 2015 low flow in the upper reaches. In Thukela Estuary the lowest oxygen content (2.8 $mg \cdot l^{-1}$) was recorded during 2015 high flow in the middle reaches while the highest (8.9 $mg \cdot l^{-1}$) was recorded during 2014 low flow. In the aMatikulu Estuary the lowest oxygen content (3.8 $mg \cdot l^{-1}$) was recorded during 2015 high flow while the highest (11.1 $mg \cdot l^{-1}$) was recorded during 2015 low flow.

Conductivity values in the uMvoti Estuary ranged from 57–486 mS/m. The lowest value was recorded from the upper reaches during 2015 high flow while the highest value was recorded from lower reaches during 2015 high flow. In the Thukela Estuary, conductivity values ranged from 43–2 619 mS/m. The lowest conductivity was recorded from the upper reaches during 2014 low flow while the highest was recorded from the upper reaches during 2015 low flow. Conductivity values in the aMatikulu Estuary ranged from 252–5 185 mS/m. The lowest conductivity was recorded from the middle reaches during 2015 low flow while the highest was recorded from the middle reaches during 2005 low flow.

In the uMvoti Estuary salinity values ranged from 0.3–2.6 during the study period. In the Thukela Estuary salinity values ranged from 0.2–16, while in the aMatikulu/Nyoni Estuary they ranged from 1.4–34.4 during the study period. Like the uMvoti and Thukela estuaries, higher salinities values in aMatikulu Estuary were recorded during the low flow sampling sessions. Salinity values increased from the upper to the lower reaches in all three estuaries studied.

In the uMvoti Estuary, pH values ranged from 6.7–7.9 while in the Thukela Estuary pH values ranged from 7.7–7.9. In the aMatikulu Estuary pH values ranged from 7.2–8.9. In the three estuaries studied, no trend in pH values along the estuarine salinity gradient was observed.

Turbidity values in the uMvoti Estuary ranged from 3.0–14.6 NTU during the study period. There was a general increase in turbidity values from the upper to the lower reaches. In the Thukela Estuary, turbidity values ranged from 28.3–874 NTU with the highest turbidity recorded during 2015 high flow. Turbidity values in the aMatikulu Estuary ranged from 1.2–15 NTU with no clear trend in turbidity values from the upper to the lower reaches.

Table 2.1: Measurement of in situ data recorded in the uMvoti, Thukela and aMatikulu estuaries during 2005, 2013, 2014 and 2015 low and high flow.

Estuary	Year	Flow	Site	Reaches	°C	O_2 (mg/l)	mS/m	Salinity	pH	NTU's
uMvoti	2005	LF	MV1	Lower	17.0	4.5	150.5	0.8	7.9	5.7
uMvoti	2005	LF	MV2	Middle	22.8	4.4	155.0	0.8	7.8	6.4
uMvoti	2005	LF	MV3	Upper	23.6	1.9	85.0	1.0	7.6	7.5
uMvoti	2005	HF	MV1	Lower	28.4	4.4	212.0	1.1	7.7	4.8
uMvoti	2005	HF	MV2	Middle	28.9	3.7	198.0	1.0	7.9	3.0
uMvoti	2005	HF	MV3	Upper	29.6	3.6	197.0	1.0	7.7	3.1
uMvoti	2013	LF	MV1	Lower	24.8	3.0	105.0	0.5	7.1	4.7
uMvoti	2013	LF	MV2	Middle	25.5	3.8	85.0	0.4	7.3	5.4
uMvoti	2013	LF	MV3	Upper	26.0	2.4	67.0	0.3	7.2	6.5
uMvoti	2014	LF	MV1	Lower	20.5	3.2	225.0	1.2	6.7	4.7
uMvoti	2014	LF	MV2	Middle	21.3	3.3	219.0	1.2	6.8	5.4
uMvoti	2014	LF	MV3	Upper	20.1	4.0	180.0	0.9	7.0	6.5
uMvoti	2015	HF	MV1	Lower	26.0	3.3	486.0	2.6	7.7	5.8
uMvoti	2015	HF	MV2	Middle	25.0	3.4	78.0	0.4	7.9	4.0
uMvoti	2015	HF	MV3	Upper	25.0	3.3	57.2	0.3	7.7	4.5
uMvoti	2015	LF	MV1	Lower	20.8	2.5	90.0	0.5	6.7	11.1
uMvoti	2015	LF	MV2	Middle	21.4	4.1	90.0	0.5	7.4	11.6
uMvoti	2015	LF	MV3	Upper	22.1	6.5	90.0	0.5	7.5	14.6
Thukela	2013	LF	TH1	Lower	23.6	6.4	922.0	5.2	7.7	51.6
Thukela	2013	LF	TH2	Middle	24.4	6.8	134.0	0.7	7.6	29.3
Thukela	2013	LF	TH3	Upper	24.0	6.8	52.8	0.3	7.6	37.0
Thukela	2014	LF	TH1	Lower	17.5	8.9	1256.0	7.2	7.5	51.6
Thukela	2014	LF	TH2	Middle	17.7	8.6	43.0	0.2	7.3	29.3
Thukela	2014	LF	TH3	Upper	18.9	6.5	43.0	0.2	7.3	43.0
Thukela	2015	HF	TH1	Lower	28.0	3.0	1411.0	8.0	7.7	874.0
Thukela	2015	HF	TH2	Middle	30.6	2.8	120.0	0.6	7.9	703.0
Thukela	2015	HF	TH3	Upper	28.0	3.0	80.0	0.4	7.7	802.0
Thukela	2015	LF	TH1	Lower	20.3	5.7	2619.0	16.0	7.6	50.6
Thukela	2015	LF	TH2	Middle	18.7	5.5	80.0	0.4	7.8	28.3
Thukela	2015	LF	TH3	Upper	21.4	7.7	80.0	0.4	7.9	38.0
aMatikulu/Nyoni	2005	LF	NY1	Lower	19.6	5.5	5171.0	34.4	8.2	6.0
aMatikulu/Nyoni	2005	LF	NY2	Midde	19.5	4.9	5185.0	34.1	8.2	7.0
aMatikulu/Nyoni	2005	LF	NY3	Middle	19.3	4.4	5020.0	33.1	8.1	6.0
aMatikulu/Nyoni	2005	LF	NY4	Upper	22.1	4.1	4486.0	29.4	7.9	15.0
aMatikulu/Nyoni	2005	HF	NY1	Lower	25.7	5.9	4760.0	31.1	8.4	7.0
aMatikulu/Nyoni	2005	HF	NY2	Midde	25.6	5.7	4790.0	32.2	8.3	3.0
aMatikulu/Nyoni	2005	HF	NY3	Middle	25.3	5.5	4760.0	31.3	8.4	4.0
aMatikulu/Nyoni	2005	HF	NY4	Upper	26.1	5.3	1700.0	10.4	8.1	6.0
aMatikulu/Nyoni	2015	HF	NY1	Lower	27.3	3.8	5040.0	33.0	8.4	1.2
aMatikulu/Nyoni	2015	HF	NY4	Upper	29.4	3.8	4340.0	28.0	8.1	7.3
aMatikulu/Nyoni	2015	LF	NY1	Lower	20.7	8.7	1256.0	7.2	8.2	6.6
aMatikulu/Nyoni	2015	LF	NY2	Midde	19.4	6.0	330.0	1.7	7.2	3.6
aMatikulu/Nyoni	2015	LF	NY3	Middle	20.7	6.2	252.0	1.4	7.4	3.7
aMatikulu/Nyoni	2015	LF	NY4	Upper	23.1	11.1	1208.0	6.9	8.9	3.0

2.3.2 Macrozoobenthos communities

The list of macrozoobenthos taxa recorded in all estuaries is presented in Appendix 1 Table A1. A total of eight macrozoobenthos taxa were recorded in the uMvoti Estuary from 10 336 individuals (Fig. 2.2). The highest number of taxa (n = 8) were recorded in the MV2 (middle reaches) and MV3 (upper reaches) during the low flow period of 2013 while the lowest number of taxa (1) was recorded during the high flow sampling period of 2005 and 2015 in the upper and lower reaches respectively (Fig. 2.2a) in this estuary. Dominant macrozoobenthos groups recorded in this estuary during the current study are presented in the appendix 1 (Fig A1). The most numerically dominant groups in the uMvoti Estuary during the current study were Insecta followed by Oligochaeta (Fig. A1). Number of taxa in the uMvoti Estuary generally increased from the upper to the lower reaches (Fig. 2.2a). A higher number of macrozoobenthos taxa were recorded during low flow compared with high flow sampling sessions (Fig. 2.2a). In comparison with the other estuaries studied, the Thukela Estuary had the highest number of macrozoobenthos taxa with a total of 24 taxa recorded from 29 589 individuals (Fig. 2.2c). All macrozoobenthos taxa (n = 24) were recorded in the TH1 (lower reaches) during the 2013 low flow. In contrast the lowest number of macrozoobenthos taxa (1) was recorded during the 2015 high flow sampling session in the upper reaches (Fig. 2.2c). The most numerically dominant group in the Thukela Estuary during the current study was Polychaeta (Fig A2). Generally, Oligochaeta was the second dominant group in this system (Fig. A2). Again, number of macrozoobenthos taxa generally increased from the upper to the lower reaches of this estuary.

A total of 11 macrozoobenthos taxa were recorded in the aMatikulu/Nyoni Estuary from 31 764 individuals (Fig. 2.2). All of these taxa (n = 11) were recorded in the NY4 (upper reaches) during the low flow period of 2015 while the lowest number was recorded in the NY2 and NY3 during low flow session of 2005 (Fig. 2.2e). The most numerically dominant groups in the aMatikulu/Nyoni Estuary during the current study were Polychaeta (2005 LF and HF), Mysida (2015 LF) and Tanaidacea (2015 HF) (Fig. A3). Number of macrozoobenthos taxa generally increased from the lower to the upper reaches (Fig. 2.2e). Unlike the other two estuaries, there was no clear pattern in number of macrozoobenthos taxa between the low and high flow sampling sessions (Fig. 2.2e).

In the uMvoti Estuary, the highest abundance of macrozoobenthos taxa was recorded in the upper reaches (10 336 no·m^{-2}) during the 2013 low flow while the lowest (11 no·m^{-2}) was recorded during the 2014 low flow sampling session in the upper reaches (Fig. 2.2b). Generally, macrozoobenthos abundance was higher during low flow compared with the high flow sampling sessions. Macrozoobenthos abundance generally increased from the lower to the upper reaches (Fig. 2.2b) in this estuary. There was a significant difference in macrozoobenthos taxa abundance of the uMvoti Estuary between sampling sites ($p < 0.05$, $F = 2.63$).

In the Thukela Estuary macrozoobenthos abundance was higher during the 2013 low flow sampling session (29 589 $no{\cdot}m^{-2}$) in the upper reaches when compared with other sampling sessions (Fig. 2.2d). The lowest abundance (3 $no{\cdot}m^{-2}$) of macrozoobenthos was recorded during the 2015 high flow sampling session in the upper reaches. Unlike the number of taxa in the Thukela Estuary, macrozoobenthos abundance increased from the lower to the upper reaches. Macrozoobenthos abundance was higher during the low flow compared with the high flow sampling sessions (Fig. 2.2d). There was a significant difference in macrozoobenthos abundance of the Thukela Estuary between the sampling sites ($p < 0.05 = F = 2.63$).

Similar to the number of taxa in the aMatikulu Estuary, macrozoobenthos abundance increased from the lower to the upper reaches. The highest macrozoobenthos abundance (31 764 $no{\cdot}m^{-2}$) was recorded during the 2015 low flow sampling session in the upper reaches while the lowest abundance (68 $no{\cdot}m^{-2}$) was recorded during the 2005 low flow sampling session in the middle reaches (Fig. 2.2f) of this estuary. There was no clear pattern in macrozoobenthos abundance along the aMatikulu/Nyoni Estuary salinity gradient. However, there was a significant difference in macrozoobenthos abundance between sampling sites ($p < 0.05$ $F = 2.63$).

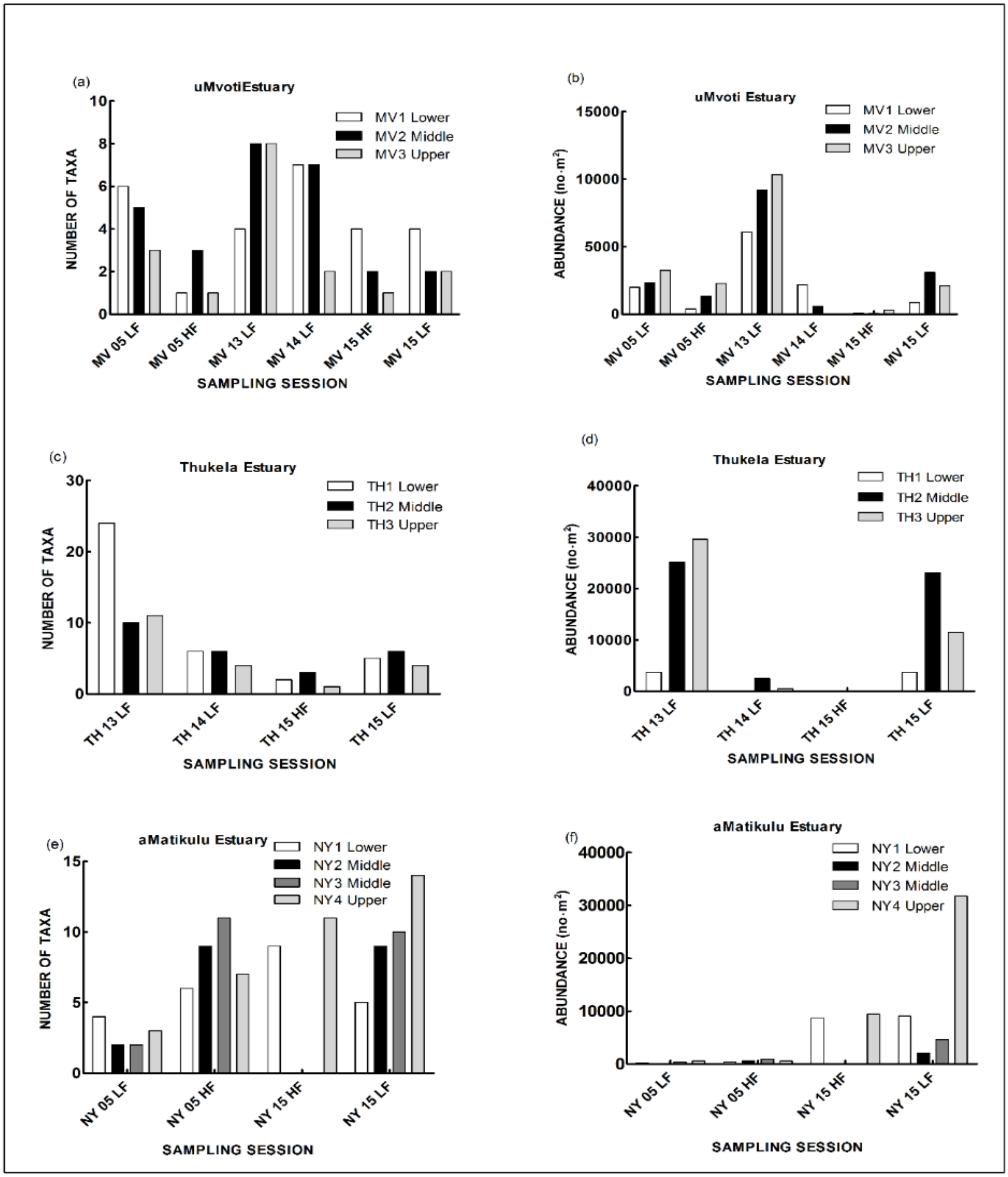

Fig. 2.2: Number of taxa (a, c and e) and abundance (b, d and f) for macrozoobenthos of the uMvoti, Thukela and aMatikulu estuaries during the study period (MV = uMvoti, TH = Thukela, NY= aMatikulu/Nyoni, HF = High flow, LF = Low flow, 05 = 2005, 13 = 2013, 14 = 2014, 15 = 2015). Data represent the mean (N = 3).

RDA tri-plots, which were constructed using log transformed species data, separated macrozoobenthos data into three distinct faunal assemblages representing the three estuaries studied (Fig. 2.3a). The triplot explained 62.2% of variation in the data (47% on axis 1 and 18% on axis 2). The uMvoti estuary was mostly dominated by freshwater macrozoobenthos taxa while the Thukela Estuary was dominated by both the estuarine and freshwater macrozoobenthos taxa. However, the aMatikulu/Nyoni Estuary was mainly dominated by macrozoobenthos taxa of marine and/ estuarine origin (Fig. 2.3a). There was a significant difference in macrozoobenthos community structure between flows ($p < 0.05$, $F = 3.48$), with both flows having a more or less equal influence on the community structure (Fig. 2.3b). There was also a significant difference in macrozoobenthos community between sampling years ($p < 0.05$, $F = 3.48$). Year 2013 low flow (LF) had more influence in structuring the macrozoobenthos community during the current study followed by year 2015 (LF) and 2005 (HF) (Fig. 2.4a). The triplot explained 54.7% of the variation in the macrozoobenthos data (33.3% from axis 1 and 21.4% from axis 2). Environmental variables responsible for structuring the macrozoobenthos community assemblages in the uMvoti, Thukela and aMatikulu/Nyoni estuaries are shown in Fig. 2.4b. The RDA triplot revealed that course sand and very course sand were the main drivers in structuring macrozoobenthos community in the uMvoti Estuary (Fig. 2.4b). In the Thukela Estuary the most important driver in structuring macrozoobenthos community was turbidity (Fig. 2.4b). Medium sand, fine sand, oxygen and salinity were the main important drivers in structuring the macrozoobenthos communities in the aMatikulu/Nyoni Estuary (Fig. 2.4b). The triplot explained 50% of the variation in the data (25.9% from axis 1 and 23.9 from axis 2).

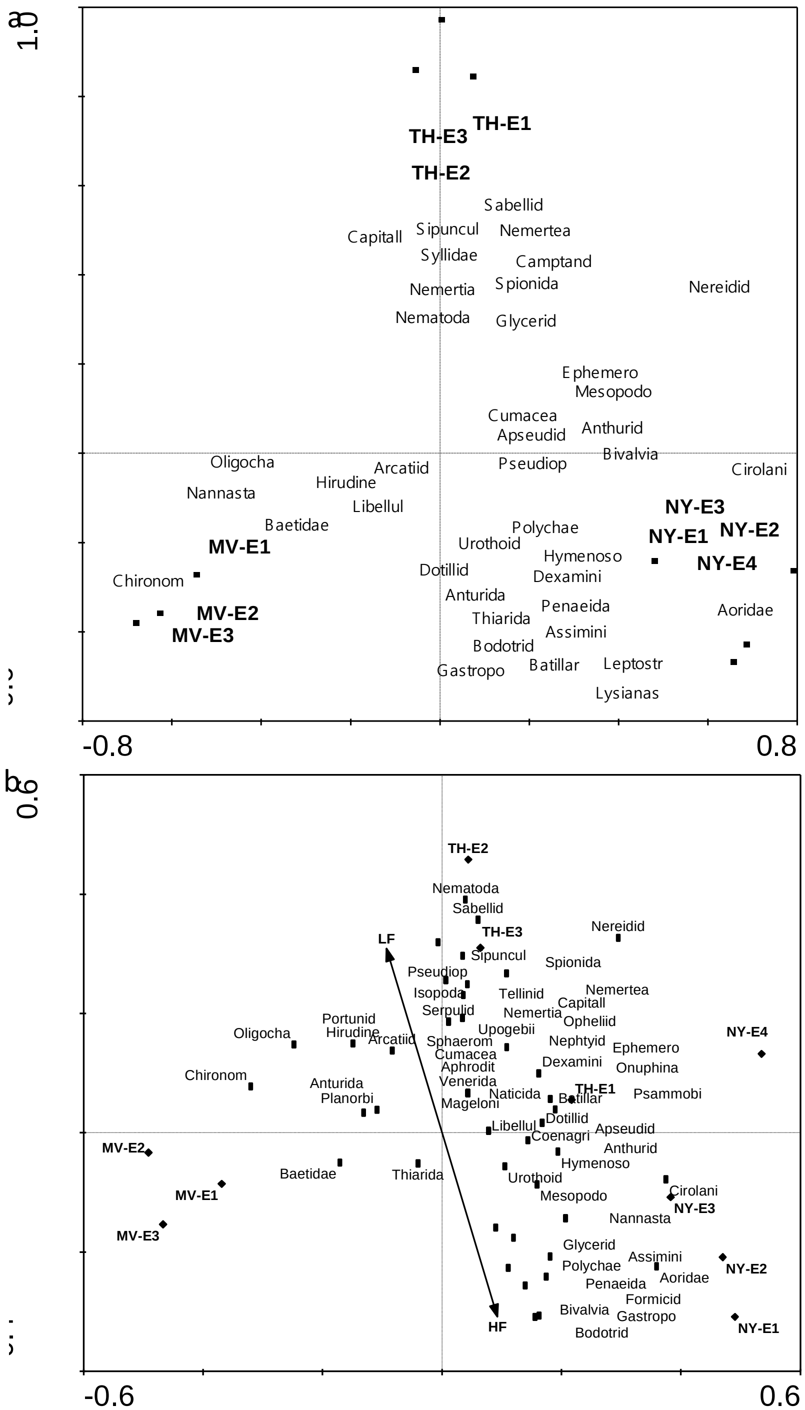

Fig. 2.3: RDA triplots showing the relationship between benthic taxa and (a) sampling sites and (b) sampling seasons. (MV-E1-3 = uMvoti Estuary site 1-3; TH-E1-3 = Thukela Estuary sites 1-3, NY- E1-4 = aMatikulu/Nyoni Estuary sites 1-4).

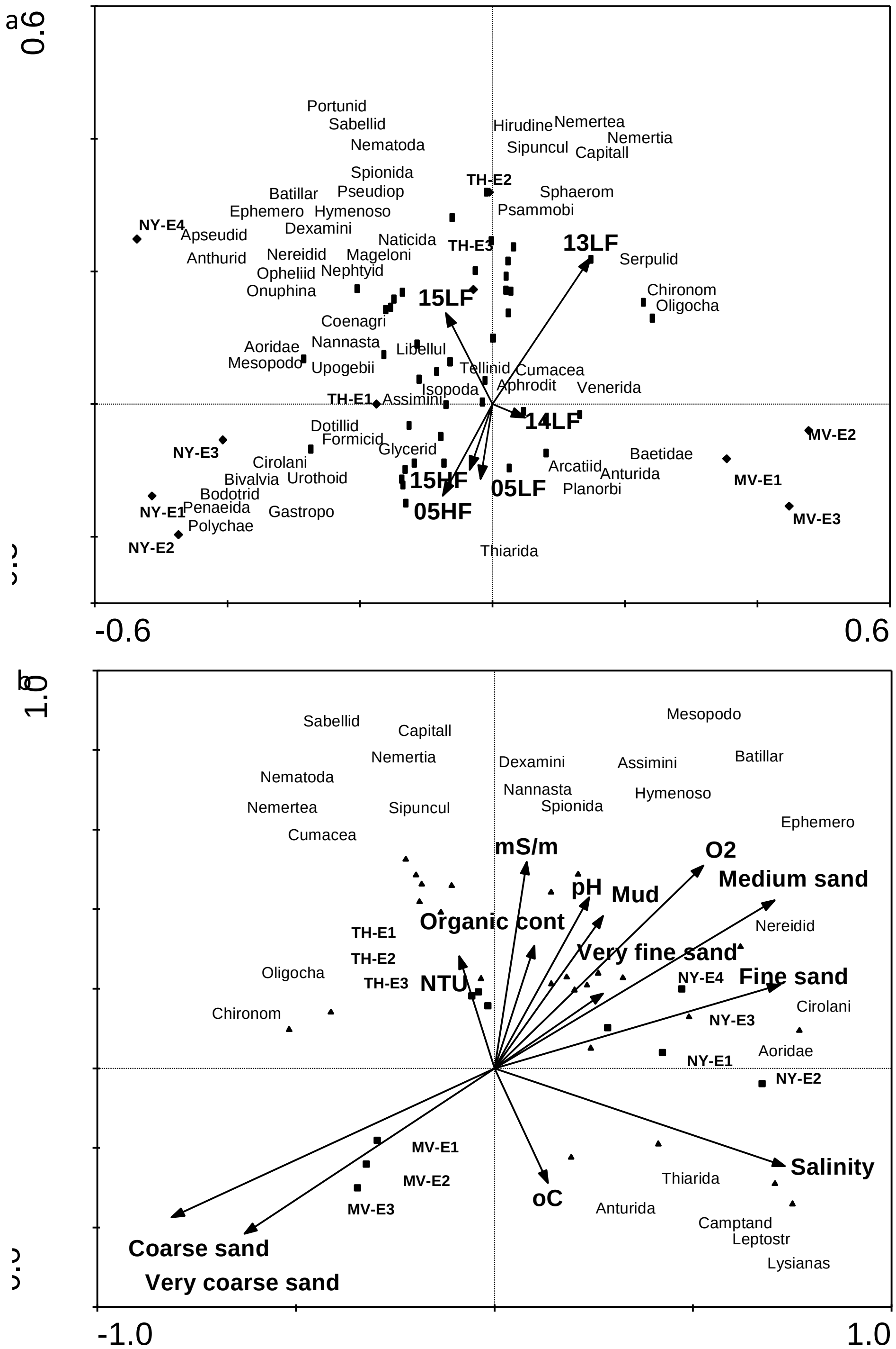

Fig. 2.4: RDA triplots showing relationship between benthic taxa and (a) sampling years and flows and (b) selected water quality variables. (MV-E1-3 = uMvoti Estuary site 1-3; TH-E1-3 = Thukela Estuary sites 1-3, NY- E1-4 = aMatikulu/Nyoni Estuary sites 1-4).

2.4 Discussion

2.4.1 Environmental variables

In the current study some of the physico-chemical parameters between the three estuaries were highly variable. However, there were no substantial differences in water temperatures between the uMvoti, Thukela and aMatikulu/Nyoni estuaries but all three estuaries experienced lower water temperatures in the lower reaches near the mouth region. The efficient flushing by the tidal inflow of colder sea water could have resulted in a decreased water temperature in the lower reaches of these three estuaries. Although the three estuaries occur in the same geographical area and are similar in form (Harrison et al. 2000), high variation in oxygen content between the three estuaries can be attributed to different anthropogenic land use activities and their varying intensities acting on these systems. The aMatikulu/Nyoni Estuary had the highest oxygen content while the uMvoti Estuary had the lowest. Generally, threats to the aMatikulu/Nyoni Estuary are relatively minimal with little anthropogenic impact on the estuary observed currently. There is only one sugar mill and associated agricultural activities upstream of the aMatikulu River (pers. obs.). However, the uMvoti Estuary is largely affected by pollution from different anthropogenic and land use sources further upstream including effluent input of treated sewage, sugar and paper mill effluents from Gledhow Sugar Mill and Sappi Stanger Mill, agricultural irrigation and several domestic uses by informal settlements (pers. obs.). Such impacts in the uMvoti Estuary are likely to have resulted in the drop in oxygen levels in this system. Following the uMvoti Estuary, the Thukela Estuary low oxygen values are likely attributed to the anthropogenic land use activities upstream which include industries, agriculture, mining, waste water treatment works and paper mill (pers. obs.). Lower oxygen levels that are the result of anthropogenic land use activities have been previously reported in the Thukela Estuary e.g. (O'Brien and Venter 2012). In addition, conductivity values differed largely between the uMvoti, Thukela and aMatikulu/Nyoni estuaries with the higher values recorded in the aMatikulu/Nyoni Estuary in our study. Compared with the uMvoti and Thukela estuaries which are largely driven by river flow, the aMatikulu/Nyoni Estuary can be controlled by the marine tidal influence. The higher salts content in the aMatikulu/Nyoni Estuary is likely to have an impact on the conductivity values of this system as supported by Pelkie et al. (1992) who reported that conductivity is dependent on the concentration of dissociated salts. This was evident in the current study since higher conductivity was recorded in the lower reaches near the mouth area where marine waters were dominant. We found no substantial variation in pH between the three estuaries although the highest pH was recorded in the aMatikulu/Nyoni Estuary. The pH values recorded during the study period were within the range that is preferred by most estuarine organisms as reported by USEPA (2006).

We found that turbidity values varied largely between the three estuaries studied with the highest turbidity values being recorded from the Thukela Estuary. The lowest turbidity values recorded from the aMatikulu/Nyoni Estuary indicated less disturbance in this system when compared with the Thukela Estuary. Generally, the higher turbidity values were recorded during high flow and this can be explained by the associated higher rainfall which results in sediment disturbance increasing the levels of total suspended solids as reported by Froneman (2002). There has been an increase in pressure on the structure and function of the Thukela system as a result of increasing demand for water related ecosystem services from the Thukela River catchment (Pienaar 2005). This was evident in the Thukela River Estuary with high load of soft sediments accumulated in the estuary with poor flushing of the river as a result of reduced flow. This soft sediment accumulation thus increased the turbidity levels in the Thukela Estuary. As historically reported, the Thukela Estuary continues to be threatened by sedimentation (Bosman et al. 2007).

We found high variability in salinity levels between the uMvoti, Thukela and aMatikulu Estuary during the study period. The lower salinity values recorded from the Thukela and uMvoti systems are attributed to these system being controlled by river discharge more than marine tidal influence. Marine water has little influence on these systems as reported by Badenhorst (1990) who concluded that the uMvoti Estuary has a limited potential for significant tidal exchange. Furthermore, Whitfield and Harrison (2003) reported that in the Thukela Estuary seawater intrusion is most effective during spring high tide when river flow is at minimum (from July to September). Salinities recorded in the three estuaries of this study were within ranges described by Whitfield (1992) for different South African estuarine types.

2.4.2 Macrozoobenthos communities

The number of macrozoobenthos taxa recorded in the uMvoti Estuary (8) was higher than that previously recorded (6) in this system (Swemmer 2009). The macrozoobenthos abundance (10 336 no·m^{-2}) that we recorded in the uMvoti Estuary was three-fold higher than that previously recorded in this system (Swemmer 2009). In the aMatikulu/Nyoni Estuary, we recorded macrozoobenthos abundance of 31 764 no·m^{-2} that was 35-fold higher than previously recorded in this system (Swemmer 2009). We found macrozoobenthos abundance was lower in the lower reaches of all the three estuaries we studied. The low abundances of species in the mouth area has been reported to be attributed from the flushing effect of high discharge from the river (Fowles 1996). Favorable habitat, good water quality together with minimal anthropogenic land use activities in the catchment of aMatikulu Estuary explained the higher abundances of macrozoobenthos in this system when compared with uMvoti and Thukela estuaries (pers. obs.). The aMatikulu/Nyoni Estuary is also probably more favorable due to a higher range or salinities and a more extensive tidal prism as well as more diverse substrate. It also does not get flushed as frequently and is therefore a more stable environment.

In the uMvoti and Thukela estuaries, we found that the number of macrozoobenthos taxa increased from the upper to the lower reaches. This trend was previously reported for southern African POEs (Day 1974a; Branch and Grindley 1979; Schlacher and Wooldridge 1996). However, the aMatikulu/Nyoni Estuary had the opposite trend with the number of taxa increasing from the lower to the upper reaches. This pattern might be attributed to the confluence of the Nyoni Estuary with the aMatikulu Estuary in the upper reaches of this system. The Nyoni Estuary might be contributing more macrozoobenthos taxa to this vicinity. Another reason that has been reported for the low number of taxa in the lower reaches is the inability of species to tolerate changes in water temperature near the mouth during ebb tides (Day 1974a). Higher fluctuations of salinity levels cause physiological stress to macrozoobenthos and may lead to lower number of taxa in the lower reaches of estuaries (Sanders et al. 1965)

Polychaeta was the most dominant taxon in terms of abundance in the Thukela and aMatikulu/Nyoni estuaries during our study. However, in the aMatikulu/Nyoni Estuary, Mysida (49%) and Tanaidacea (38%) were the most dominant taxa during March 2015 and August 2015 respectively. Such changes in dominant groups with river flow conditions has been reported in South African estuaries because of the dynamic nature of community change in the benthos (Wooldridge and Deyzel 2009b). Similar to the Thukela and aMatikulu/Nyoni estuaries, Polychaeta were reported to be dominant in the Gamtoos and Mfolozi–Msunduzi Estuary during high flow period (Schlacher and Wooldridge 1996; Ngqulana et al. 2010). As in the current study, the dominance of chironomids and oligochaetes was previously reported in the uMvoti

Estuary (Swemmer 2009). Chironomids, which are mainly freshwater species, were reported to be indicative of organically polluted freshwater systems (Rae 1989). Again the water quality of the uMvoti system was described as grossly polluted since 1964 (Begg 1978). This may also explain the dominance of the insects Chironomidae in the uMvoti Estuary. In addition, chironomid larvae and oligochaete worms are good indicators of pollutants as they are resistant to higher levels of perturbation (Day 1981a, Hawking and Smith 1997).

The uMvoti Estuary catchment is heavily affected by the anthropogenic water resource use activities as described above. The impacts of these activities was reflected in the water quality of this system which in turn is reflected in the macrozoobenthos community comprising low diversities and abundances. Tharme (1996) stated that the uMvoti riverine system has been modified completely, with nearly total loss of natural habitat and biota and the destruction of many basic ecosystem functions. These impacts on the river are also reflected in the uMvoti Estuary. As a result, the uMvoti Estuary is regarded as a degraded system which functions differently from the way it did in its former pristine state (MacKay et al. 2000). Chironomidae and Oligochaeta are the dominant benthic fauna of non-relict estuarine lakes (Allanson et al. 1990). Similarly, the dominance of chironomids and oligochaetes in the Gamtoos Estuary occurred during low salinity conditions (Schlacher and Wooldridge 1996). Similar to the current study, a reduction in macrozoobenthos abundance following high flow occured in other South African POEs (e.g. Gamtoos and Great Berg estuaries) (Schlacher and Wooldridge 1996). Findings from their studies suggested a negative effect of high freshwater inflow on macrozoobenthos abundance. Schlacher and Wooldridge (1996) concluded that a drop in salinity was likely to have contributed to the reduction of macrozoobenthos abundances although this was not concluded from causal evidence but from correlative one.

Although a method of determining estuarine zones based on salinity tolerance of biological organisms has been developed (Bulger et al. 1993) and although Day's (1967a, 1981b) classification system is considered important by Australian and South African estuarine researchers (De Villiers and Hodgson 1999; Jackson and Jones 1999), it has been reported that there could be an improvement in these systems through addition of sediment characteristics as an additional factor influencing the distribution of macrozoobenthos (Jackson and Jones 1999). Canonical analyses of the present study revealed that oxygen, salinity, medium and fine sand had the highest influence in structuring the macrozoobenthos community of the aMatikulu/Nyoni

Estuary. Mud content and pH were the second most important variables which structured the macrozoobenthos community in the aMatikulu/Nyoni Estuary. Similarly, salinity, oxygen and pH were the environmental variables responsible for structuring the macrozoobenthos community in the aMatikulu/Nyoni Estuary (Swemmer 2009; Venter 2013). Dissolved oxygen was a responsible variable in structuring the macrozoobenthos community in the Mfolozi–Msunduzi Estuary (Ngqulana et al. 2010) as found in the aMatikulu/Nyoni Estuary in the current study. A previous study by Swemmer (2009) also showed that aMatikulu/Nyoni Estuary had higher species richness and abundance than the uMvoti Estuary. It was reported that aMatikulu/Nyoni Estuary had more favorable habitat conditions (i.e. the mud, fine and medium sized composition of the substratum and organic content of higher percentage (Swemmer 2009)). These favorable habitat conditions were also apparent during the current study. In addition to the relatively good water quality conditions such favorable habitat conditions are suggested to have contributed to higher species richness and higher abundances in the aMatikulu/Nyoni Estuary during the current study. In the uMvoti Estuary course and very course sand were the most important variables in structuring the macrozoobenthos community. The dominance of Oligochaeta and Chironomidae in the uMvoti Estuary were previously attributed to the estuary being shallow and freshwater driven together with medium to course sand substratum (Wepener and MacKay 2002). Course and very course sand also played a role in structuring macrozoobenthos communities in the uMvoti Estuary during the present study. In the Thukela Estuary, turbidity was the most important environmental variable driving macrozoobenthos community. Turbidity has been previously reported to be the environmental variable structuring the macrozoobenthos community in the uMvoti Estuary (Swemmer 2009). River inflow, together with tidal exchange, enhances turbidity gradients in estuaries which are important during the nursery phase of some marine species through provision of olfactory cues for juveniles (Allanson and Read 1995).

Salinity is described as the key variable determining distribution patterns of macrozoobenthos in estuaries of South Africa e.g. (Teske and Wooldridge 2003; Wooldridge and Deyzel 2009a), North America e.g. (Holland et al. 1987; Pollac et al. 2011), Europe e.g. (Ysebaert et al. 2003; Mariano and Barros 2014). Many estuarine organisms survive periods of adverse salinity levels, although the effects on the distribution of these species may occur if these conditions persist for longer periods (Teske and Wooldridge 2003). However, if changes in salinity are drastic and occur rapidly, the effect of salinity will override substratum preferences for most invertebrates (Teske and Wooldridge 2003). The macrozoobenthos of the Thukela Estuary was dominated by Oligochaetes and freshwater taxa such as chironomids insects (MacKay et al. 2004). Similarly, freshwater insects such as Chironomids dominated the Thukela Estuary during August 2014 in the current study. Oligochaetes also dominated this system in 2013 and 2015 during low and high flow respectively. The alteration of water quality as a result of several anthropogenic land use activities in the upper catchment of Thukela Estuary might have also contributed to the dominance of pollution tolerant chironomid insects in this system. The dominance of chironomids (associated with low salinities) was reported in the lower reaches of some Eastern Cape estuaries during the dry season (Teske and Wooldridge 2003). In 2001 the macrozoobenthos of the Thukela Estuary displayed a polychaete-dominated community during low flow (MacKay et al. 2004). Similarly, we found that polychaetes dominated the Thukela Estuary during 2013 and 2015 irrespective of flow. This could be because of more gentle salinity gradient and more stable system during low flow. The Thukela Estuary is a part of a riverine system draining a relatively large catchment compared with the other river dominated estuaries, and it generally carries larger volumes of freshwater so saltwater generally only intrudes 2 km above the mouth (MacKay et al. 2004).

Historically, water quality variables like temperature, oxygen, nutrient and salt loads have been a concern in the Thukela system (Venter 2013). Some water quality constituents previously recorded in the uMvoti River have a negative impact on the structure and functioning of this system, and historically, such constituents have been of serious concern (Venter 2013). Although there is an increased pressure on the structure and function of the Thukela system as a result of increasing demand for anthropogenic water related ecosystem services, the lower portion of the Thukela River and its associated estuary are characterized as an ecologically important region of the Thukela catchment. Water quality of the lower aMatikulu River was reported to be in fairly good/ slightly modified state while the state of the lower Thukela River and Estuary were reported as moderately modified (Class C) (IWR 2004).

Although the three estuaries studied here are from the same geographical region and are similar in form, the high variability and differences in their water quality and macrozoobenthos communities were mostly explained by different anthropogenic land use activities in their respective catchments. Although some of the land use activities are common in the uMvoti and

Thukela estuaries (e.g. effluent discharges, agriculture and water abstraction), the intensity and other water resource use activities e.g. industries, might be the determinant of the varying water quality states and macrozoobenthos community structures in these three estuaries.

As mentioned the anthropogenic land use activities that are associated with the three estuaries studied differed. We showed that macroinvertebrates communities respond to stressors associated with land use activities. Alteration of water quality and quantity as a result of land use activities was evident in the present study. In the case of uMvoti Estuary, excessive use of water resources for anthropogenic activities is likely to have resulted in significant changes in the macroinvertebrate community. This may be associated with disrupted ecosystem processes and potentially the functioning of this system. High loads of soft sediments in the Thukela Estuary, coupled with reduced flow is concerning. This may be affecting the productivity of the system specifically in relation to limited sunlight penetration as well as reduced nutrient concentrations. Management intervention is urgently needed for these estuarine systems. Estuary Management Plans need to be developed and implemented for the protection of these systems in KZN with the Thukela Estuary being the priority followed by aMatikulu/Nyoni Estuary. The Thukela Estuary is an important system with high ecological value. This system displayed higher species richness than the other two estuaries studied and it has been acknowledged for its nursery function. With so many stressors acting in its catchment, the management measures in this system are needed to protect its diversity and ecological functioning. The aMatikulu/Nyoni Estuary is relatively pristine with higher species richness and abundances and lies within the nature reserve. However, an increase in sediment load suggests that new stressors are acting on its catchment. This estuarine system needs to be conserved together with the upper river catchment to prevent stressors from affecting the community structures and functioning of this estuarine system. The uMvoti Estuary is heavily impacted with a shift in communities, loss of biodiversity and nursery function. Although this system is heavily threatened and polluted, management measures are needed so as to improve water quality and biodiversity of this estuary and also to regain the nursery function of this system. If these three estuaries are not managed or conserved, they will continue to deteriorate as a result of land use activities in their catchments. This may lead to a complete loss of nursery function as already seen in uMvoti Estuary, loss of biodiversity and severely altered water quality.

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2.7 APPENDIX 1

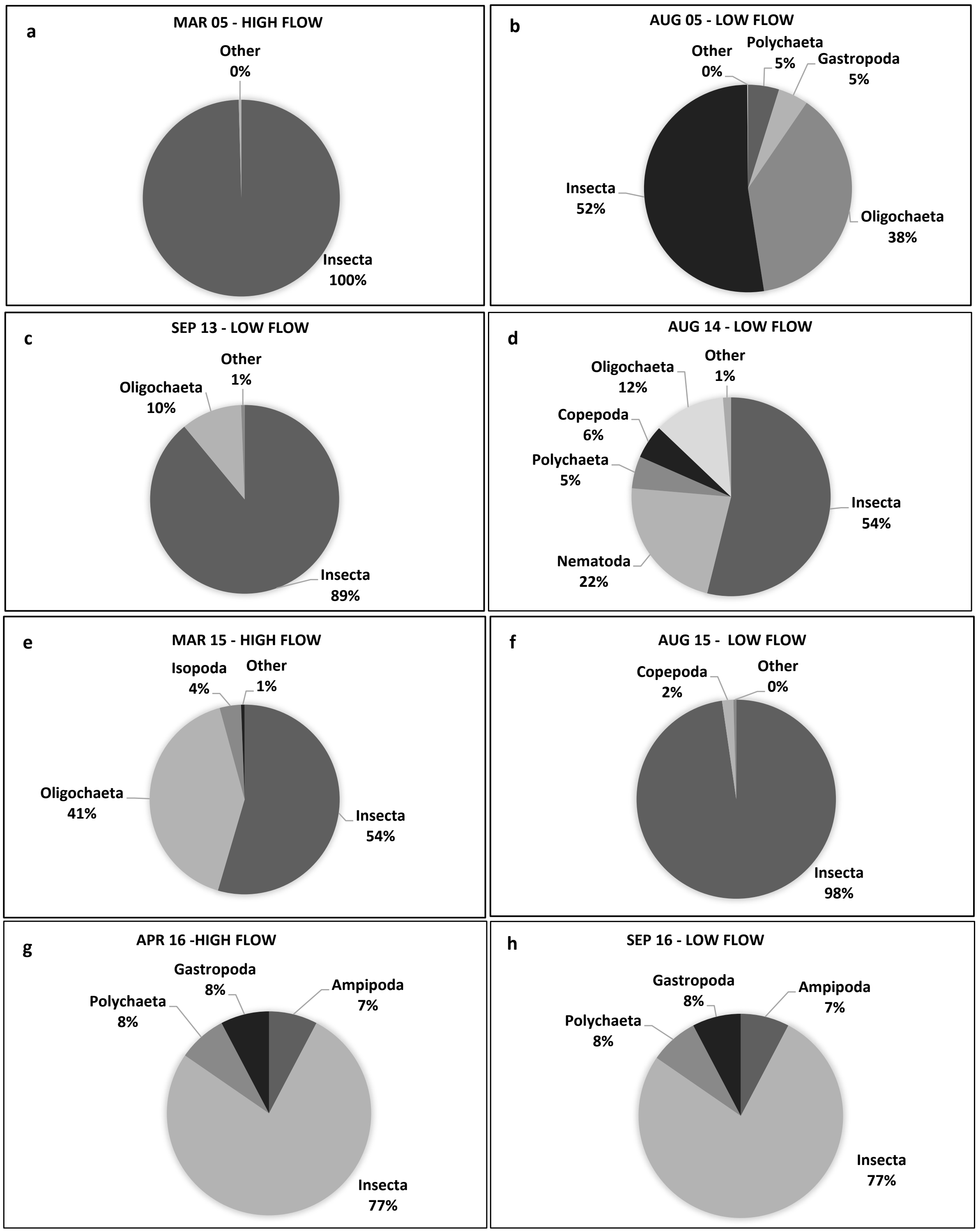

Fig. A1: Dominant taxonomic groups recorded in the uMvoti Estuary during high and low flow sessions of the study period. All taxa which contributed less than 2% of the total abundance were grouped together as "Other".

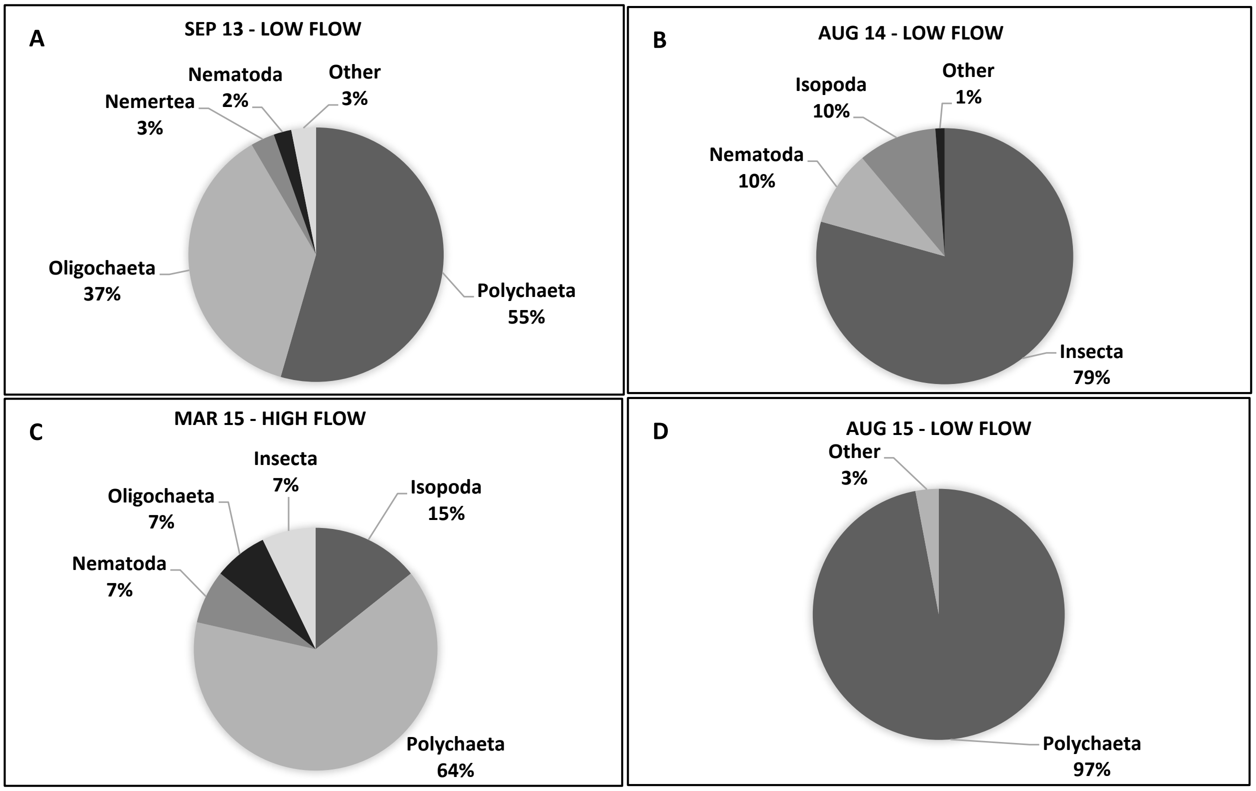

Fig. A2: Dominant taxonomic groups recorded in the Thukela Estuary during high and low flow sessions of the study period. All taxa which contributed less than 2% of the total abundance were grouped together as "Other".

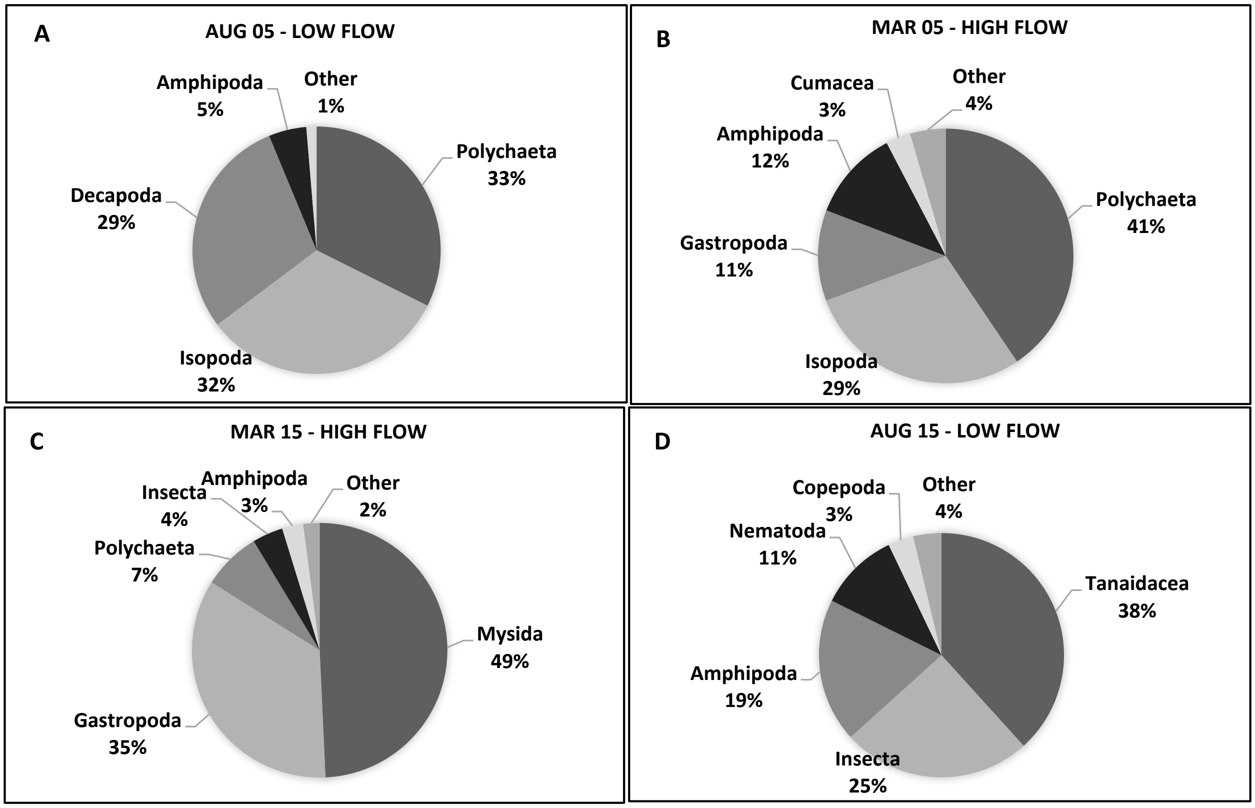

Fig. A3: Dominant taxonomic groups recorded in the aMatikulu/Nyoni Estuary during high and low flow sessions of the study period. All taxa which contributed less than 2% of the total abundance were grouped together as "Other".

Table A1: List of taxa and abundances (no·m^{-2}) recorded in the uMvoti, Thukela and aMatikulu estuaries during high and low flow sessions of the study period. Codes on the table represent the estuary e.g. MV=uMvoti, TH = Thukela, NY=aMatikulu/Nyoni Estuary; year e.g. 05=2005; flow e.g. L= low flow and site e.g.E1=site 1.

TAXA	NY05L-E1	NY05L-E2	NY05L-E3	NY05L-E4	NY05H-E1	NY05H-E2	NY05H-E3	NY05H-E4	NY 15H-E1	NY15H-E4	NY15L-E1	NY15L-E2	NY15L-E3	NY15L-E4	MV05L-E1	MV05L-E2	MV05L-E3	MV05H-E1	MV05H-E2	MV05H-E3	MV13L-E1	MV13L-E2	MV13L-E3	MV14L-E1	MV14L-E2	MV14L-E3	MV15H-E1	MV15H-E2	MV15H-E3	MV15L-E1	MV15L-E2	MV15L-E3	TH13L-E1	TH13L-E2	TH13L-E3	TH14L-E1	TH14L-E2	TH14L-E3	TH15H-E1	TH15H-E2	TH15H-E3	TH15L-E1	TH15L-E2	TH15L-E3
ANNELIDA																																												
Aphroditidae																																	13											
Capitallidae																																	13		51									
Hirudinea sp.																					38	25	25																				28	
Magelonidae																																	25											
Nephtyidae																																	165											
Nereididae	119		85	153	42	356	237	280	833	500		403		3750	331	34																	13	6802	13566		205			25		3403	22375	11319
Oligochaeta sp.								34		14		14			1068	1678	119		17		88	1799	773	306	14		6	22	161			14	25	16897	5231	3				3				
Opheliidae																																	89											
Onuphinae																																	13											
Polychaeta sp.	25	34			59																																							
Sabellidae												14																					355	13	9196								42	14
Sipunculidae																																	114	76	532									
Spionidae										14		375	194	2764										128	14	3							761	440	444									
Serpulidae																																			76									
MOLLUSCA																																												
Assiminidae									431																								13											
Batillaridae										5875		83		222																														
Bivalvia sp.						8	17																																					
Gastropoda sp.					42	68	153	8							25																													
Naticidae																																	13											
Planorbidae																																												
Psammobiidae																																	139											
Tellinidae																																					11							
Thiaridae							8								322																													
Veneridae																																	13											
ARTHROPODA																																												
Aoridae	63				34	8	153	8	389	42	1153	903	56	5361																														
Arcatiidae																									131											3	3							
Apseudidae													15319	15319																														
Anthuridae													333											8	3																			
Bodotridae					51	17																																						
Baetidae															8				17			13																						
Cirolanidae	25	34	153	203	153	34	288	220	194		56			14										3			17						507			33	44	161	3		3	153		
Chironomidae									333	222	6028		2861	1056	237	586	3119	381	1322	2271	5953	7232	9525	1108	381		58	36	156	764	3056	2097		101	89	3	2247	14	3			42	69	56
Cumacea sp.																																	13											
Coenagrionidae										28			14																															
Dexaminidae										56				153																														
Dotillidae			153	203		17																																						
Ephemeroptera sp.										97			14	14																														
Formicidae									14																																			
Hymenosomatidae									14	14		222	56	69																14														
Isopoda sp.																																						44						
Libellulidae													14																															
Mysidae									6375	2583														22									203			14						69		
Nannastacidae									153		361	28		14																														
Penaeidae	17					8	8																																					
Pseudiopdomidae													14	1319											22					69	42												597	97
Portunidae																																		13										
Sphaeromatidae																																	38											
Upogebiidae				17																																								
Urothoidae	51	17	8																											14														
NEMERTEA																																												
Nemertea sp.																																	444	38	13									
NEMATODA																																												
Nematoda sp.											1472		1375	1375								139	13	594	19	8							747	418	177		6	267		3		14	14	
Total density	300	85	399	576	381	516	864	550	8736	9444	9069	2042	20250	31431	1991	2298	3238	381	1356	2271	6079	9208	10336	2169	583	11	81	58	317	861	3097	2111	3714	24798	29375	55	2516	486	6	31	3	3681	23125	11486
Number of taxa	6	4	4	4	6	8	7	5	9	11	5	8	11	14	6	3	2	1	3	1	3	5	4	7	7	2	3	2	2	4	2	2	21	9	10	5	6	4	2	3	1	5	6	4

CHAPTER 3

Response of zooplankton communities to altered water quality and seasonal flow changes in selected river dominated estuaries in KwaZulu-Natal, South Africa

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Running header: Response of zooplankton communities in river dominated estuaries

Abstract

Globally, estuaries are among the most ecologically important and productive ecosystems with many threatened by anthropogenic activities. Zooplankton is used as a bioindicator of anthropogenic impacts and ecosystem integrity. The spatial and temporal composition of zooplankton communities were quantified and compared within and between three estuaries (uMvoti, Thukela and aMatikulu/Nyoni estuaries) with different anthropogenic land uses in KwaZulu-Natal, South Africa. Additional effects of physico-chemical variables and seasonal flow patterns to zooplankton community structuring were analyzed.

Zooplankton samples were collected between August 2014 and September 2016. Associated water samples were also collected for water quality analyses in the laboratory.

Ten taxa were identified in each of the Thukela and aMatikulu/Nyoni estuaries while a total of five were recorded in the uMvoti Estuary. Highest zooplankton abundance was recorded in the aMatikulu/Nyoni Estuary (15086.9 ind. m^{-3}) followed by Thukela (973 ind. m^{-3}) and then uMvoti Estuary (456 ind. m^{-3}). Redundancy analysis (RDA) plots revealed higher salinity, conductivity and oxygen as determinants in structuring zooplankton community in the aMatikulu/Nyoni Estuary while turbidity and pH were determinants in structuring the zooplankton communities in the uMvoti and Thukela estuaries.

Elevated concentrations of DIN in the Thukela Estuary during high flow identifies the Thukela River as an important source of nitrogen to this estuary. Such flow dependent patterns highlight the importance of adequate release policy and adherence to this policy for this heavily utilized estuary. This study revealed that the estuaries studied must be managed to ensure sufficient freshwater supply which controls primary production which in turn is food source for zooplankton. Although the three estuaries were from the same biogeographical region with similar river dominated function, high variability in their zooplankton communities and water quality could be explained by differing anthropogenic land use activities in their catchments.

Keywords:

Anthropogenic land use; community structure; estuaries; flows; pollution; water quality; zooplankton

3.1 Introduction

Ever increasing human population settling near estuaries increases pressure on these systems and make them more susceptible to external perturbations such as pollution (Nicolas et al., 2007; Perissinotto et al., 2010). The ecological value of these interface ecotones of freshwater and marine systems includes the unique processes and diversities present and the provision of nursery grounds for many marine species (Vasconcelos et al., 2010). Physical and chemical parameters in estuaries fluctuate rapidly both spatially and temporarily and such fluctuations may be experienced by organisms when moving against vertical and horizontal axes of the environment (Kibirige & Perissinotto, 2003). Zooplankton distribution in estuaries varies both temporarily and spatially as a result of highly dynamic conditions experiences in these systems (Schlacher & Wooldridge, 1994). Zooplankton community structures are also negatively affected by climate change (Hansen et al., 1988). Zooplankton serves as an important food source for many fish species (Whitfield 1985, 1998). These organisms play a significant role in energy transfer from primary producers to secondary production (Wooldridge & Bailey, 1982; Harrison & Whitfield 1990). Zooplankton also serve as a good indicator of biodiversity as well as environmental change (Fahd et al., 2007).

Zooplankton communities in South African permanently open estuaries (POE) are mostly dominated by copepods and mysids (Wooldridge, 1999). In South Africa, estuaries along the south-west coast are dominated by copepods *Acartia longipatella* and *A. africana* while those in the east coast are dominated by *A. natalensis* and *Pseudodiaptomus charteri* (Grindley, 1981; Wooldridge, 1999). However, the copepod *P. hessei* is recorded in almost all South African estuaries (Wooldridge, 1999).

Estuarine systems in many parts of the world exhibit taxonomically structured zooplankton communities (Lee & McAlice, 1979; Greenwood, 1981; Ambler et al., 1985). In South African estuaries zooplankton communities are structured by the freshwater input and the estuary mouth states (Montoya-Maya & Strydom, 2009). Generally, zooplankton biomass is related to riverine inflow into the estuary. The effect of altered river flow in Kariega and Great Fish estuaries was investigated and zooplankton biomass correlated with freshwater input (Grange et al., 2000). Furthermore, temporal patterns of zooplankton abundance are related to frequency of freshwater impulse into the estuarine system and do not follow seasonal cycle (Wooldridge, 1999).

Globally there has been a serious deterioration in the ecological health of many estuaries as a result of excessive water abstraction, agricultural activities and industrial effluents, (Owens, 1991; Kennish, 2002; Quinton & Catt, 2007; Zhang et al., 2012). These systems receive major anthropogenic input that is generally generated further upstream from point and non-point sources and from urban areas and industries near them (Chapman & Wang, 2000). Estuaries are dependent on riverine freshwater input for primary production (Kimmerer, 2002). In South Africa, approximately 59 % of estuaries are not protected (Van Niekerk & Turpie, 2012). About 54 % of estuaries along the South African coastline are described as in a fair to poor state (Van Niekerk & Turpie, 2012). Many estuaries along the north coast of KwaZulu-Natal (KZN) Province are affected by the reduced flows, poor water quality, and habitat alterations originating from anthropogenic land use activities (King & Pienaar, 2011). Mitigation opportunities for these estuaries are limited due to the lack of understanding of how these systems respond to these impacts. The understanding of how threats associated with multiple land use activities impact on the ecological structure, function and processes of ecosystems in KZN in relation to conservation and management requirements (endpoints) is urgently needed.

The structure, function, processes and biodiversity of estuarine ecosystems are vulnerable to threats from local and catchment scale land use activities (Stewart-Koster et al., 2010, O'Brien & Wepener, 2012). Anthropogenic pressure and impacts on estuarine systems are intense because most urbanization is concentrated in coastal areas (Almeida et al. 2012). The magnitude of pollution sources like sewage outfalls and aquaculture effluents together with natural environmental variability make some coastal systems to vary greatly in terms of levels of pollution, eutrophication and disturbance (Almeida et al., 2012). The uMvoti Estuary is rated severely degraded in terms of sedimentology and this condition deteriorates with time due to high sediment loads during flooding conditions (Badenhorst, 1990). The uMvoti Estuary is regarded as a polluted system. The Thukela Estuary provides habitat for some marine migrants, estuarine and freshwater species, and acts as a conduit for many anadromous species that populate the middle and upper reaches of the Thukela Estuary (Whiitfield & Harrison, 2003). Another ecological importance of the Thukela Estuary is the transportation of sediment into the local marine ecosystem which is linked to the ecological functioning of a biodiversity hot-spot in a near shore marine environment and the commercially important Thukela Banks Fisheries (Bosman et al., 2007; Lamberth & Turpie, 2003). There has been a deterioration in the ecological health of the Thukela Estuary and the lower Thukela River, near shore marine and Thukela Banks ecosystems over the last few decades (Lamberth et al., 2009). The impacts and deterioration are linked to changes in land use within the catchment and alteration in volume, timing and duration of flows by abstraction to meet the demand of users (Lamberth et al., 2009). The aMatikulu/Nyoni Estuary is considered to be in a good condition although siltation from the catchment is of concern (Whitfield, 2000). As a consequence of its good condition the aMatikulu/Nyoni Estuary was selected as the reference site for the present study.

In this study the spatial and temporal composition of zooplankton communities was quantified and compared within and between the three estuaries (uMvoti, Thukela and aMatikulu/Nyoni estuaries) in KZN with different catchment anthropogenic land use activities. Additional effects of physico-chemical variables and seasonal flow patterns to zooplankton community structuring were analyzed. The three estuaries studied are similar in function, and geographical area. We hypothesized that there will be spatio-temporal variation in zooplankton communities within and between the three estuaries and that the differences would be related to variations in catchment land use patterns and water quality.

3.2 Methods

3.2.1 Study areas

The three estuaries (uMvoti, Thukela and aMatikulu/Nyoni) (Chapter 2) on the KZN north coast, South Africa are river dominated (Whitfield 2000). The uMvoti and Thukela estuaries are considered as river mouths while the aMatikulu/Nyoni Estuary is considered as a permanently open estuary (Whitfield 2000) (Fig. 2.1). The uMvoti Estuary (29°23’ S, 31°20’ E) (Chapter 2) is a subtropical river mouth situated north of the coastal town of KwaDukuza (Stanger). The estuary occupies a catchment area of 0.2 km^2. The uMvoti River catchment is subject to agricultural activities which include commercial forestry, sugar cane farming, commercial dry land agriculture and subsistence farming. The uMvoti Estuary has a substantial recreational value (O’Brien et al. 2009). The Thukela Estuary (29°13’ S, 31°29’ E) (Chapter 2) is a subtropical river mouth with a surface area of approximately 0.6 km^2 (Begg 1978; Whitfield 2000). The principal physical process in the estuary is the river discharge and as a result the estuary mouth is usually open (IWR Environmental 2003). The ecological importance of the Thukela Estuary include transportation of sediment into the local marine ecosystem which is linked to the ecological functioning of a biodiversity hot-spot in a near shore marine environment and the commercially important Thukela Banks Fisheries (DWAF 2004). The Thukela Estuary act as a conduit for many anadromous species thriving in the upper reaches and this system can serve as a habitat for some estuarine resident, marine migrants and freshwater fish species (DWAF 2004). The aMatikulu/Nyoni Estuary (36°06’36”S, 31°37’09”E) (Chapter 2) is a subtropical permanently open estuary which covers a catchment area of 900 km^2 (Begg 1978; Whitfield 2000). The lower reaches of the aMatikulu/Nyoni Estuary are disturbed by agricultural activities but the fauna is generally in a good condition (Harrison et al. 2000). This estuarine system forms part of the aMatikulu Nature Reserve, managed by Ezemvelo KwaZulu-Natal Wildlife (EKZNW).

3.2.2 Sampling and laboratory analysis.

Zooplankton samples were collected in the uMvoti, Thukela and aMatikulu/Nyoni Estuary during 2014 (August), 2015 (March and August) and 2016 (April and September). Sampling dates were selected so that they represented high flow (March and April) as well as low flow (August and September) as referred to hereafter. Historical data for uMvoti and Thukela zooplankton samples collected in 2014 were also incorporated to the data set during analyses. Three sites were sampled in uMvoti and Thukela estuaries and four sites in aMatikulu/Nyoni Estuary. *In situ* physico-chemical data including oxygen, salinity, temperature, pH and turbidity were recorded during each survey using a calibrated portable meter (Eutech instruments CyberScan series 600, Thermo Fisher, USA). Water quality samples for further analyses in the laboratory were collected from the subsurface using polyethylene bottles and were sent to Umgeni Water Laboratory (an accredited laboratory with the South African National Accreditation System and the International Standard ISO/IEC 17025:2005) for nutrients (dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP)) and chlorophyll a analysis. Zooplankton samples were collected in each estuary during daytime. A 200 µm mesh plankton net attached to a hyperbenthic sled was used to collect samples from the sediment/water interface of the estuary. The sled was allowed to settle at the bottom of the estuary and was towed for 20 m before it was retrieved. Three replicate samples were collected at each site during all sampling sessions. All zooplankton samples were preserved in 10% formalin containing Rose

Bengal dye. During March and August 2015 sampling, high reed density and grass in the middle reaches (NY2 and NY3 in March 2015 and NY3 in August 2015) of aMatikulu/Nyoni Estuary prevented boat access. As a result, some gaps in data from these two middle sites in the aMatikulu/Nyoni Estuary during this year occur. Although some water quality data from the same sites and dates were obtained from the Department of Water and Sanitation, South Africa. Data for DIN, DIP and Chl–a are absent for April 2016 in the uMvoti Estuary because of insufficient sample volume.

In the laboratory, samples were diluted to 1–5 L solutions depending on the concentration of the sample. Organisms in each sample were kept in suspension by thorough stirring of the sample and three sub samples for identification and enumeration withdrawn using a 20 ml scoop after penetrating the entire depth, while stirring continuously to prevent settlement of organisms (Perissinotto and Wooldridge, 1989; Jerling and Wooldridge, 1995). The coefficient of variation between subsamples was below 10%. Organisms in each subsample were identified to the lowest taxa possible and enumerated using a dissecting microscope and zooplankton abundance expressed as the number of individuals per cubic meter ($ind \cdot m^{-3}$).

3.2.3 Statistical analysis

Ordination techniques were applied using the original zooplankton community data sets which allowed for intepretation of zooplankton community structures with regard to taxa recorded during the study (Van den Brink et al., 2003). These practises evaluated changes in zooplankton community structures and then tested the statistical significance of differences in communities after incorporated with Monte Carlo permutation testing (Van den Brink et al., 2003; Ter Braak and Smillauer, 2004). Initially, the technique articulated zooplankton taxa between sampling sites and sampling sessions without incorporating environmental data and this allowed for evaluations of spatio-temporal composition of zooplankton community (Ter Braak, 1994). The largest part of the total variance of the data was used to establish the first latent variable and the second latent variable was established using the remaining variance in the data set (Van den Brink et al., 2003). The ordination diagram with two axes was created using these two latent variables. Sites and taxa were firstly presented as points at the location of the values in the diagram. Samples with comparable taxa composition lay close to each other while those with differing taxa composition were located far apart (Van den Brink et al., 2003). After incorporating available environmental data, tri-plots were constructed. These tri-plots displayed arrows of environmental data which were directed to higher values where there was existence of correlations between sites and environmental variables (Van den Brink et al., 2003). Redundancy analysis (RDA) using the CANOCO version 4.5 software was the linear response that was adopted to accomplish this (Ter Braak, 1994). Because zooplankton abundance data were available, the data were transformed using a Log X+2 - transformation (Van den Brink et al., 2003). Redundancy analyses were also performed to detect if there were any significant differences in zooplankton communities between sites, years and flows.

3.3 Results

3.3.1 Environmental variables

Measurements of the physico–chemical variables including temperature, dissolved oxygen, salinity, turbidity and chlorophyll a (Chl–a) recorded during the current study are presented in Table 3.1. In the current study water temperature exhibited lower values during the low flows. Horizontally, lower temperature values were generally measured in the lower reaches in the three estuaries studied. Temporally, water temperature in the uMvoti Estuary ranged from a minimum of 20.1 ^{o}C during August 2014 to a maximum 29.1 ^{o}C during April 2016. In the Thukela Estuary the lowest water temperature (17.5 ^{o}C) was recorded during August 2014 while the highest (30.6 ^{o}C) was recorded during March 2015. Temporally, water temperature in the aMatikulu/Nyoni Estuary ranged from a minimum of 20.7 ^{o}C during August 2015 to a maximum of 29.4 ^{o}C during March 2015. Horizontally, dissolved oxygen concentrations in the uMvoti Estuary increased from the lower to the upper reaches. The lowest oxygen concentration (1.8 mg l^{-1}) was recorded during April 2016 in the middle reaches while the highest (6.5 mg l^{-1}) was recorded during August 2015 in the upper reaches. Temporally, dissolved oxygen concentrations in the Thukela Estuary ranged from a minimum of 2.8 mg l^{-1} during March 2015 in the middle reaches to a maximum 8.9 mg l^{-1} during August 2014 in the lower reaches. Contrary to the uMvoti Estuary, dissolved oxygen concentration in the aMatikulu/Nyoni Estuary generally increased from the upper to the lower reaches. In this estuary the lowest dissolved oxygen concentration (3.7 mg l^{-1}) was recorded during April 2016 in the upper reaches while the highest (9.8 mg l^{-1}) was recorded during September 2016 in the lower reaches.

Surface salinity values in the uMvoti Estuary ranged from a minimum of 0.5 during the 2015 high flow to a maximum of 1.6 during the 2016 high flow. Although there was a general increase in salinity values from the upper to the lower reaches, salinity values were the same throughout the estuary during August 2014 and March 2015. In the Thukela Estuary, salinity values ranged from a minimum of 0.3 during the 2015 low flow to a maximum of 63.6 during the 2016 high flow. Horizontally, no clear pattern in salinity values was observed in the Thukela Estuary during the current study. Temporally, salinity values in the aMatikulu/Nyoni Estuary ranged from a minimum of 1.4 during the 2015 high flow to a maximum of 53.9 during the 2016 low flow. Along the estuary, salinity levels increased from the upper to the lower reaches. Turbidity levels in the uMvoti Estuary ranged from a minimum of 3 NTU during the 2015 high flow to a maximum of 14.6 NTU during the 2015 low flow with no clear trend in turbidity values along the estuary. In the Thukela Estuary, lowest turbidity (28 NTU) was recorded during the 2014 low flow while the highest (875 NTU) was recorded during the 2016 high flow. Horizontally, turbidity levels increased from the upper to the lower reaches. Turbidity levels in the aMatikulu/Nyoni Estuary ranged from a minimum of 1 NTU during the 2016 high flow to a maximum of 7.3 NTU during the 2015 high flow with no clear trend in turbidity values along the estuary.

Table 3.1: Measurements of temperature, oxygen, salinity, turbidity and chlorophyll a in the uMvoti, Thukela and aMatikulu/Nyoni estuaries during the study period. LF = low flow, HF = high flow, MV1-3 = uMvoti Estuary sites 1-3; TH1-3 = Thukela Estuary sites 1-3, NY1-4 = aMatikulu/Nyoni Estuary sites 1-4.

Estuary	Year	Flow	Site	Reaches	^{o}C	O_2 ($mg \cdot l^{-1}$)	Salinity	NTU	Chl-a ($\mu g \cdot l^{-1}$)
uMvoti	2014	LF	MV1	Lower	20.5	3.2	0.9	11.1	*
uMvoti	2014	LF	MV2	Middle	21.3	3.3	0.9	11.6	*
uMvoti	2014	LF	MV3	Upper	20.1	4.0	0.9	14.6	*
uMvoti	2015	HF	MV1	Lower	26.0	3.3	0.5	4.8	0.5
uMvoti	2015	HF	MV2	Middle	25.0	3.4	0.5	3.0	0.5
uMvoti	2015	HF	MV3	Upper	25.0	3.3	0.5	4.1	0.5
uMvoti	2015	LF	MV1	Lower	20.8	2.5	0.9	12.1	43.2
uMvoti	2015	LF	MV2	Middle	21.4	4.1	0.9	13.6	42.5
uMvoti	2015	LF	MV3	Upper	22.1	6.5	0.9	14.6	66.4
uMvoti	2016	HF	MV1	Lower	29.1	2.3	1.6	5.8	0.3
uMvoti	2016	HF	MV2	Middle	25.4	1.8	1.4	3.0	0.3
uMvoti	2016	HF	MV3	Upper	25.3	4.8	1.1	4.4	0.3
uMvoti	2016	LF	MV1	Lower	25.8	1.8	0.9	12.0	*
uMvoti	2016	LF	MV2	Middle	26.0	2.2	0.9	10.0	*
uMvoti	2016	LF	MV3	Upper	26.4	2.3	0.9	9.0	*
Thukela	2014	LF	TH1	Lower	17.5	8.9	0.3	50.6	*
Thukela	2014	LF	TH2	Middle	17.7	8.6	0.7	28.3	*
Thukela	2014	LF	TH3	Upper	18.9	6.5	0.8	38.0	*
Thukela	2015	HF	TH1	Lower	28.0	3.0	0.5	874.0	0.5
Thukela	2015	HF	TH2	Middle	30.6	2.8	0.5	703.0	0.5
Thukela	2015	HF	TH3	Upper	28.0	3.0	0.5	802.0	0.5
Thukela	2015	LF	TH1	Lower	20.3	5.7	0.3	51.6	13.7
Thukela	2015	LF	TH2	Middle	18.7	5.5	0.7	29.3	5.3
Thukela	2015	LF	TH3	Upper	21.4	7.7	0.8	38.0	5.1
Thukela	2016	HF	TH1	Lower	25.4	5.4	49.5	875.0	0.3
Thukela	2016	HF	TH2	Middle	26.5	7.2	29.2	708.0	0.3
Thukela	2016	HF	TH3	Upper	23.6	5.1	63.6	802.0	0.3
Thukela	2016	LF	TH1	Lower	24.0	5.0	8.5	65.0	*
Thukela	2016	LF	TH2	Middle	26.8	5.9	3.3	30.0	*
Thukela	2016	LF	TH3	Upper	26.9	3.7	2.7	40.0	*
aMatikulu/Nyoni	2015	HF	NY1	Lower	27.3	3.8	5.0	1.2	15.8
aMatikulu/Nyoni	2015	HF	NY4	Upper	29.4	3.8	5.0	7.3	8.7
aMatikulu/Nyoni	2015	LF	NY1	Lower	20.7	8.7	7.2	6.6	0.5
aMatikulu/Nyoni	2015	LF	NY3	Middle	20.7	6.2	1.4	3.7	4.0
aMatikulu/Nyoni	2015	LF	NY4	Upper	20.6	6.4	1.5	3.7	3.1
aMatikulu/Nyoni	2016	HF	NY1	Lower	25.7	6.2	33.6	1.0	0.3
aMatikulu/Nyoni	2016	HF	NY2	Middle	26.3	6.7	30.6	6.0	0.3
aMatikulu/Nyoni	2016	HF	NY3	Middle	27.9	5.7	28.7	4.0	0.3
aMatikulu/Nyoni	2016	HF	NY4	Upper	26.1	3.7	29.9	4.0	0.3
aMatikulu/Nyoni	2016	LF	NY1	Lower	24.5	9.8	53.9	6.6	*
aMatikulu/Nyoni	2016	LF	NY2	Middle	23.8	7.9	36.9	4.0	*
aMatikulu/Nyoni	2016	LF	NY3	Middle	25.9	8.5	27.3	3.7	*
aMatikulu/Nyoni	2016	LF	NY4	Upper	23.3	5.5	28.0	4.0	*

* absent chlorophyll a values

In Table 3.1 absent chl–a values in the uMvoti, Thukela and aMatikulu/Nyoni estuaries as a result of sample size are represented by (*) symbol. Temporally, maximum pelagic chl–a values in the uMvoti Estuary were measured during the low flow when compared with the high flow. Chlorophyll a concentrations in the uMvoti Estuary ranged from a minimum of 0.3 µg l^{-1} during 2016 to a maximum of 66.4 µg l^{-1} during 2015. Although there was an increase in chl–a values in the uMvoti Estuary from the lower to the upper reaches during the 2015 low flow, chl–a values during 2015 and 2016 high flows were the same along the estuary. Similar to the uMvoti Estuary, maximum chl–a values in the Thukela Estuary were measured during the low flow. Lowest chl–a value (0.3 µg l^{-1}) in the Thukela Estuary was measured during 2016 high flow while the highest (13 µg l^{-1}) was measured during 2015 low flow. Although there was an increase in chl–a values from the lower to the upper reaches during the 2015 low flow, chl–a values were generally the same along the Thukela Estuary during the 2015 and 2016 high flows. Contrary to the uMvoti and Thukela estuaries, no clear pattern in chl–a values along the estuary was observed in the aMatikulu/Nyoni Estuary during the current study. Temporally, pelagic chl–a values in the aMatikulu/Nyoni Estuary ranged from a minimum of 0.3 µg l^{-1} during 2016 high flow to a maximum of 15.8 µg l^{-1} during 2015 high flow. Dissolved inorganic nitrogen (DIN) concentrations were higher during the low flow (August and September) when compared with the high flow (March and April) in the uMvoti and aMatikulu/Nyoni estuaries while an opposite pattern was observed in the Thukela Estuary (Fig. 3.1). Concentrations of DIN ranged from 0.14 mg l^{-1} (2015 high flow) to 0.68 mg l^{-1} (2016 low flow) in the uMvoti Estuary, 0.01 mg l^{-1} (2015 low flow) to 0.45 mg l^{-1} (2015 high flow) in the Thukela Estuary, and 0.01 mg l^{-1} (2015 low flow) to 6.5 mg l^{-1} (2016 low flow) in the aMatikulu/Nyoni Estuary (Fig. 3.1). In the uMvoti Estuary highest dissolved inorganic phosphorus (DIP) concentration (0.05 mg l^{-1}) was recorded during 2015 low flow while the lowest (0.01 mg l^{-1}) was recorded during 2016 low flow (Fig. 3.1). Horizontally, DIP concentrations were generally decreasing from the lower to the upper reaches except for 2016 low flow where an opposite pattern was observed (Fig. 3.1). Highest DIP concentration (0.01 mg l^{-1}) in the Thukela Estuary was recorded during 2016 low flow while the lowest (0.004 mg l^{-1}) was recorded during 2015 low and high flow as well as 2016 high flow (Fig. 3.1). Horizontally, no clear trends were observed in the DIP concentrations of the Thukela Estuary during the current study (Fig. 3.1). In the aMatikulu/Nyoni Estuary highest DIP concentration (0.03 mg l^{-1}) was recorded during 2016 low flow while the lowest (0.004 mg l^{-1})

was recorded during all the sampling sessions of the current study (Fig. 3.1). Highest DIP concentrations were generally recorded in the middle reaches (NY3) which is the site in the lower Nyoni system.

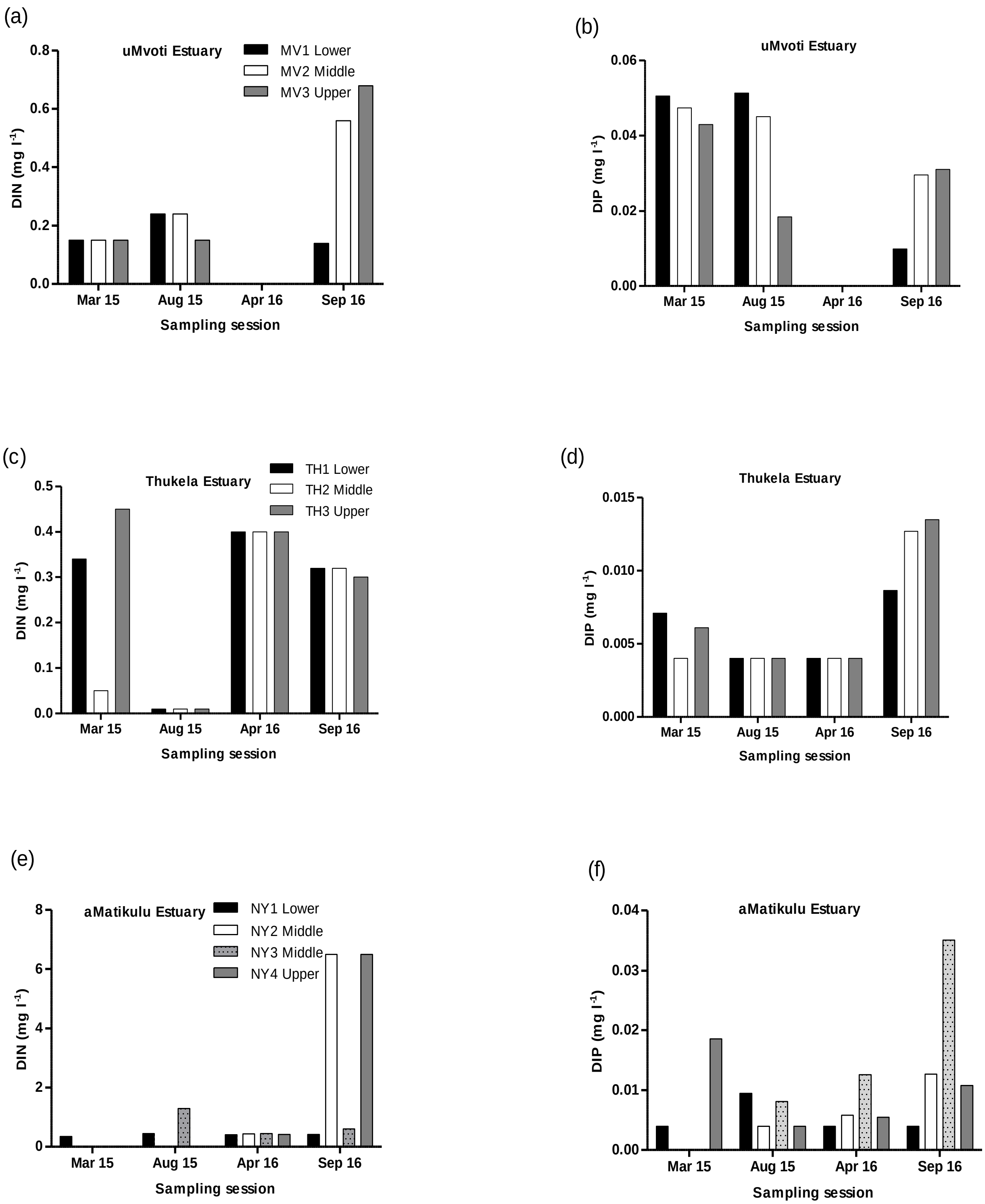

Fig. 3.1: Dissolved inorganic nitrogen (DIN) (a, c and e) and dissolved inorganic phosphorus (DIP) (b, d and f) measured in the uMvoti, Thukela and aMatikulu/Nyoni Estuary during the study period.

3.3.2 Zooplankton

Temporally, the number of zooplankton taxa in the uMvoti Estuary ranged from a minimum of 1 during the 2016 high flow to a maximum of 5 during 2014 (low flow), 2015 (low flow) and 2016 (low flow) (Table 3.2). In the Thukela Estuary the lowest number of zooplankton taxa (2) was recorded during the 2015 low flow while the highest (10) was recorded during the 2016 high flow (Table 3.3). Similar to the Thukela Estuary, number of zooplankton taxa in the aMatikulu/Nyoni Estuary ranged from 2–10 with lowest number of taxa recorded during the 2015 low flow and the highest during the 2016 low flow (Table 3.4). Zooplankton abundances in the uMvoti and Thukela estuaries were higher during the low flow compared with the high flow, however, an opposite pattern was observed in the aMatikulu/Nyoni Estuary (Fig. 3.2).

During the current study, the aMatikulu/Nyoni Estuary exhibited the highest zooplankton abundance followed by Thukela and then uMvoti Estuary (Fig. 3.2). Zooplankton abundance in the uMvoti Estuary ranged from a minimum of 2.8 ind. m^{-3} during the 2015 high flow to a maximum of 456.5 ind. m^{-3} during the 2014 low flow (Fig. 3.2a). The most dominant zooplankton taxa in the uMvoti Estuary were Acartiidae (particularly *Acartia natalensis*), Pseudodiaptomidae (particularly *Pseudodiaptomus hessei*) and Chironomidae (Fig. 3.2b). Combined the dominant zooplankton species accounted for 98% (Aug-14), 23% (Mar-15), 84% (Aug 15), 100% (Apr-16) and 61% (Sep-16) of the total zooplankton abundance in this system. Abundance for the most dominant zooplankton species in the uMvoti Estuary ranged from 1.3 to 145.3 ind. m^{-3} (Fig. 3.2b). Lowest zooplankton abundance (2.7 ind. m^{-3}) in the Thukela Estuary was recorded during the 2015 high flow while the highest abundance (955.3 ind. m^{-3}) was recorded during the 2016 low flow (Fig 3.2c). The most dominant zooplankton taxa in the Thukela Estuary were Acartiidae (particularly *A. natalensis*), Pseudodiaptomidae (particularly *P. hessei*) and Nematoda (Fig. 3.2d). Combined, these zooplankton species accounted for 55% (Aug-14), 33% (Mar-15), 94% (Aug-15), 64% (Apr-16), and 48 % (Sep-16) of the total zooplankton abundance on this system. Abundance for the most dominant zooplankton species ranged from 1.3 to 227.9 ind. m^{-3} (Fig. 3.2d). Zooplankton abundance in the aMatikulu/Nyoni Estuary ranged from a minimum of 17.5 ind. m^{-3} during the 2015 low flow to a maximum of 15086.9 ind. m^{-3} during the 2016 high flow (Fig. 3.2e). The most dominant zooplankton taxa in the aMatikulu/Nyoni Estuary during the present study were Acartiidae (particularly *A. natalensis)*, Pseudodiaptomidae (particularly *P. hessei*) and Mysidae (particularly *Mesopodopsis* *africanus*) (Fig. 3.2f). Combined these zooplankton species accounted for 99.5% (Mar-15), 95% (Aug-15), 87% (Apr-16) and 45% (Sep-16) of the total zooplankton in this estuary. Abundance for the most dominant zooplankton species ranged from 8.3 to 2049.3 ind. m^{-3} (Fig. 3.2f).

Table 3.2: Zooplankton taxa and mean abundance (Ind· m^{-3}) recorded in the uMvoti Estuary from August 2014 to September 2016.

	Aug-14			**Mar-15**			**Aug-15**			**Apr-16**			**Sep-16**		
	MV1	MV2	MV3	MV1	MV2	MV3	MV1	MV2	MV3	MV1	MV2	MV3	MV1	MV2	MV3
TAXA															
HEXANAUPLIA															
Acartiidae	45.7	4.2	264.2	0.9			44.3								
Pseudodiaptomidae	376.3	1.4	55.3				9.2		5.5	9.2			15.0	11.1	0.9
BRANCHIOPODA															
Cladocera sp.	18.0		2.8												
MALACOSTRACA															
Mysidae	1.4			131.0											
Cumacea sp.															
OSTRACODA					0.9		2.8								
Ostracoda sp.													13.0	4.6	1.4
INSECTA															
Chironomidae	15.2		6.9	1.8	1.8		48.0	12.0	108.2	18.5		9.2	30.0	24.9	14.8
Culicidae				0.9											
Insecta sp.															5.5
SCYPHOZOA															
Cyaneidae			1.4										1.4		
CLITELLATA															
Oligochaeta sp.							0.9								
NEMATODA															
Nematoda sp.								1.8	50.8					12.9	5.5
TOTAL	**456.5**	**5.5**	**330.6**	**134.6**	**2.8**	**0.0**	**155.9**	**13.8**	**164.5**	**27.7**	**0.0**	**9.2**	**59.4**	**53.5**	**22.6**
NO. OF TAXA	**5**	**2**	**5**	**4**	**2**	**0**	**5**	**2**	**3**	**2**	**0**	**1**	**4**	**4**	**5**

Table 3.3: Zooplankton number of taxa and mean abundance (Ind· m^{-3}) recorded in the Thukela Estuary from August 2014 to September 2016.

	Aug-14			Mar-15			Aug-15			Apr-16			Sep-16		
	TH1	TH2	TH3	TH1	TH2	TH3	TH1	TH2	TH3	TH1	TH2	TH3	TH1	TH2	TH3
TAXA															
HEXANAUPLIA															
Acartiidae	11.1	22.1	22.1			2.8	173.6	1.8	2.8			21.2			
Pseudodiaptomidae		6.9	1.4			0.9	126.6	81.2	114.4	9.2		34.1	21.2	349.5	459.3
Copepod sp.								0.9				0.9			
Copepod nauplii	1.4		1.4									1.8			
BRANCHIOPODA															
Cladocera sp.	513.2	2.8						0.9							
MALACOSTRACA															
Mysidae											0.9	107.0			
Cumacea sp.															
Shrimp larvae					0.9					0.9					
Penaeidae						0.9							0.9	0.9	
Luciferidae											0.9				
Aoridae												2.8			
Hymenosomidae												1.8			
OSTRACODA															
Ostracoda sp.									28.6						
INSECTA															
Chironomidae		4.2			0.9	0.9							22.1	374.4	468.5
Culicidae					0.9										
Ectinosomatidae										0.9					
Diptera sp.											0.9				
Insect sp.														1.8	
SCYPHOZOA															
Cyaneidae										0.9					
CLITELLATA															
Oligochaeta sp.													0.9	4.6	
NEMATODA															
Nematoda sp.								45.2	36.0	0.9	16.6	45.2	12.9	46.0	14.6
LEPTOCARDII															
Branchiostomatidae			4.2												
SAGITTOIDEA															
Sagittidae								1.8	1.8						
POLYCHAETA															
Nereididae										2.8				3.7	7.4
Sabellidae										0.9					
Phyllodocidae												1.8			
GASTROPODA															
Gastropod sp.													0.9		
TOTAL	**525.7**	**36.0**	**29.1**	**0**	**2.8**	**5.5**	**300.3**	**131.9**	**183.5**	**18.4**	**19.4**	**216.7**	**59.0**	**781.0**	**949.7**
NO. OF TAXA	**4**	**4**	**4**	**0**	**3**	**4**	**2**	**6**	**5**	**7**	**4**	**10**	**6**	**7**	**4**

Table 3.4: Zooplankton taxa and mean abundance (Ind· m^{-3}) recorded in the aMatikulu/Nyoni Estuary from March 2015 to September 2016.

	Mar-15				**Aug-15**				**Apr-16**				**Sep-16**			
	NY1	NY2	NY3	NY4	NY1	NY2	NY3	NY4	NY1	NY2	NY3	NY4	NY1	NY2	NY3	NY4
TAXA																
HEXANAUPLIA																
Acartiidae	40.6			11.1			202.0		20.3	7349.3	490.0	299.7		12.9	2.8	11.0
Pseudodiaptomidae				16.6	15.7		6.5		18.4	2865.8	143.3	94.1	603.1	41.5	10.1	12.0
MALACOSTRACA																
Mysidae	79.2			747.2					2.8	4132.3	1100.1	2.8				
Cumacea sp.																
Aoridae				0.9	0.9				2.8	18.5	4.6		19.4		22.1	12.0
Dexaminidae									0.9							
Leptostraca sp.													56.0	78.4	2.8	54.8
Shrimp larvae													0.9			0.9
Isopod sp.																0.9
INSECTA																
Chironomidae					0.9								5.5	0.9	0.9	5.5.
NEMATODA																
Nematoda sp.									12.0	27.7			0.9		0.9	0.9
POLYCHAETA																
Nereididae										4.6			8.3			7.4
Spionidae												3.7		0.9		
Polychaete sp.												0.9				
Phyllodosidae																2.8
TOTAL	**119.8**			**775.8**	**17.5**		**208.5**		**57.2**	**14426.0**	**1738.0**	**401.2**	**694.2**	**134.6**	**17.6**	**102.7**
NO. OF TAXA	**2**			**4**	**3**		**2**		**6**	**7**	**4**	**5**	**7**	**5**	**6**	**10**

The RDA tri-plot which was constructed using log transformed species data separated zooplankton data into three distinct faunal assemblages representing the three estuaries we studied (Fig. 3.3a). The triplot explained 76.5 % of variation in the data (65.7 % on axis 1 and 10.8 % on axis 2). There was a significant difference in zooplankton community structures between sampling sites ($p < 0.05$). Year 2016 had the highest influence in structuring the zooplankton community followed by year 2014 and then year 2015 (Fig. 3.4b). This triplot explained 73.1 % of variation in the data (51.4 % on axis 1 and 21.7 % on axis 2). There was a significant difference in zooplankton community structure between years ($p < 0.05$). There was also a significant difference between flows ($p < 0.05$), although with both flows having a more or less equal influence on the community structure (Fig. 3.4a). Water quality variables responsible for structuring the zooplankton community assemblages in the uMvoti, Thukela and aMatikulu/Nyoni estuaries are shown in Fig. 3.3b. The RDA plot showed that higher salinity, conductivity and oxygen contributed to the structuring of the zooplankton community in the aMatikulu/Nyoni Estuary while turbidity and pH contributed to the structuring of zooplankton community in the uMvoti and Thukela estuaries (Fig. 3.3b). The triplot explained 66 % of variation in the data (42.2 % on axis 1 and 23.8 % on axis 2). The influence of water quality variables in structuring the zooplankton assemblages was significant ($p < 0.05$).

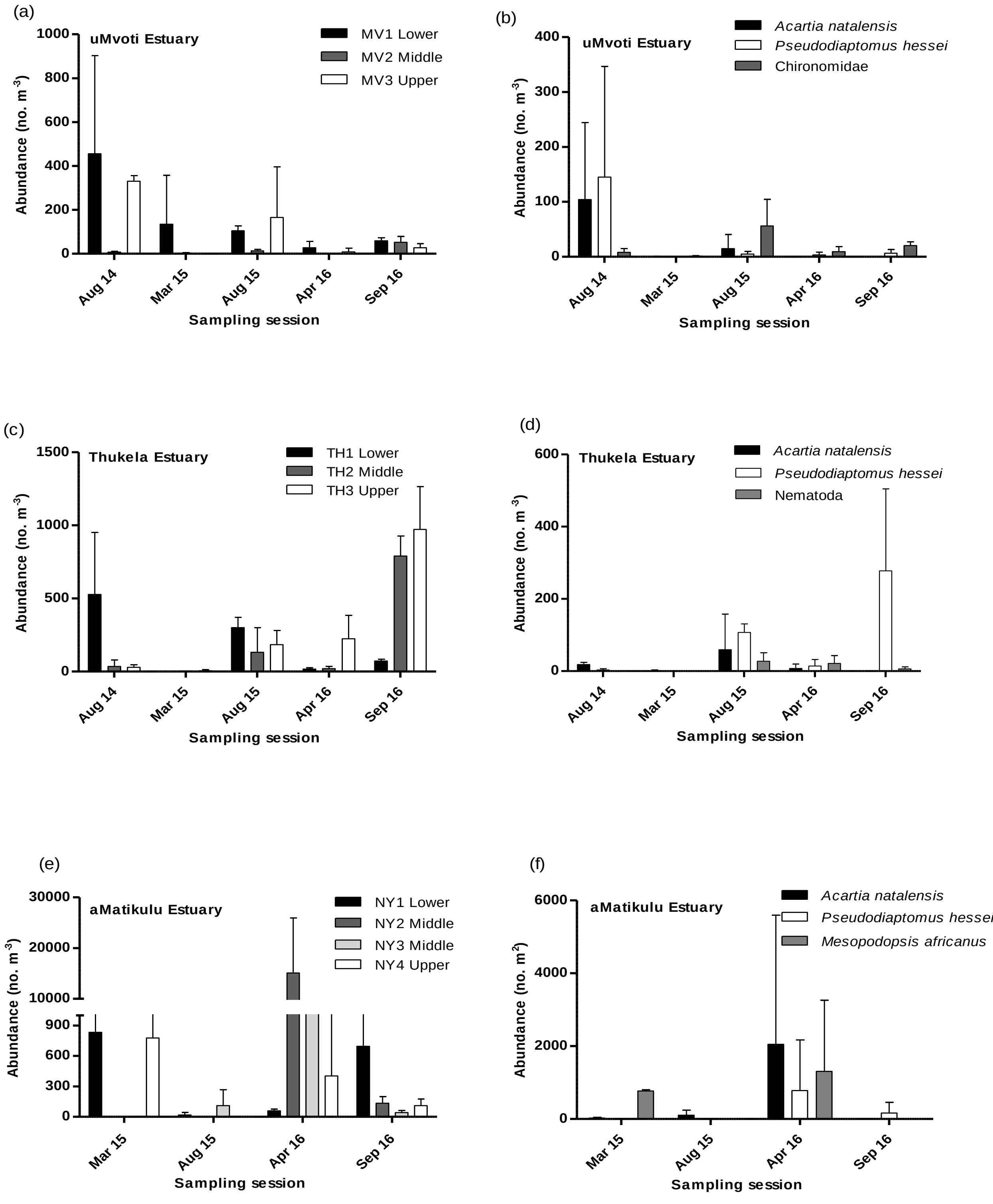

Fig. 3.2: Numerical abundance (mean ± SD, n = 3) of total zooplankton (a, c and e) and the three dominant species (b, d and f) in the uMvoti, Thukela and aMatikulu/Nyoni Estuary.

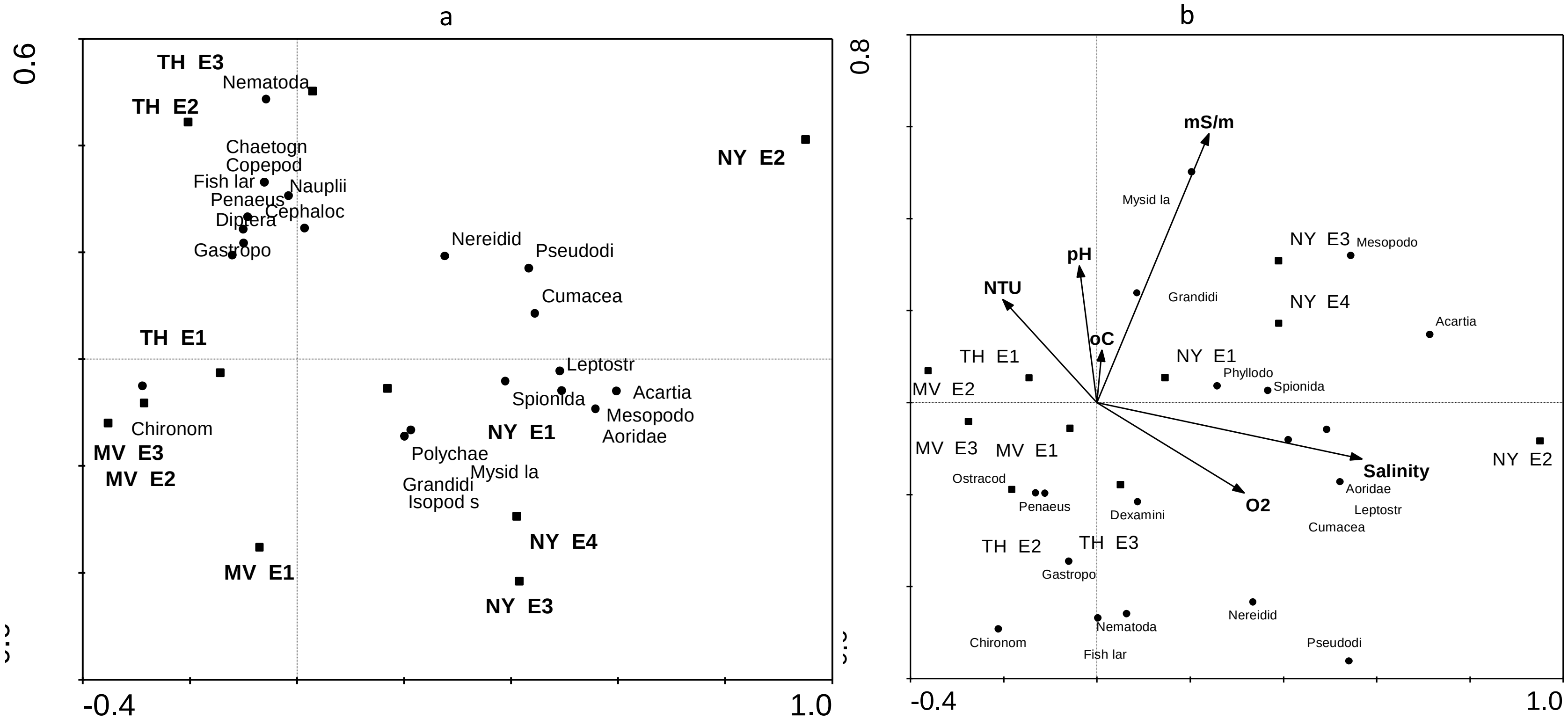

Fig. 3.3: RDA triplots showing the relationship between zooplankton species and (a) sampling sites and (b) selected water quality variables. (MV-E1-3 = uMvoti Estuary site 1-3; TH-E1-3 = Thukela Estuary sites 1-3, NY- E1-4 = aMatikulu/Nyoni Estuary sites 1-4).

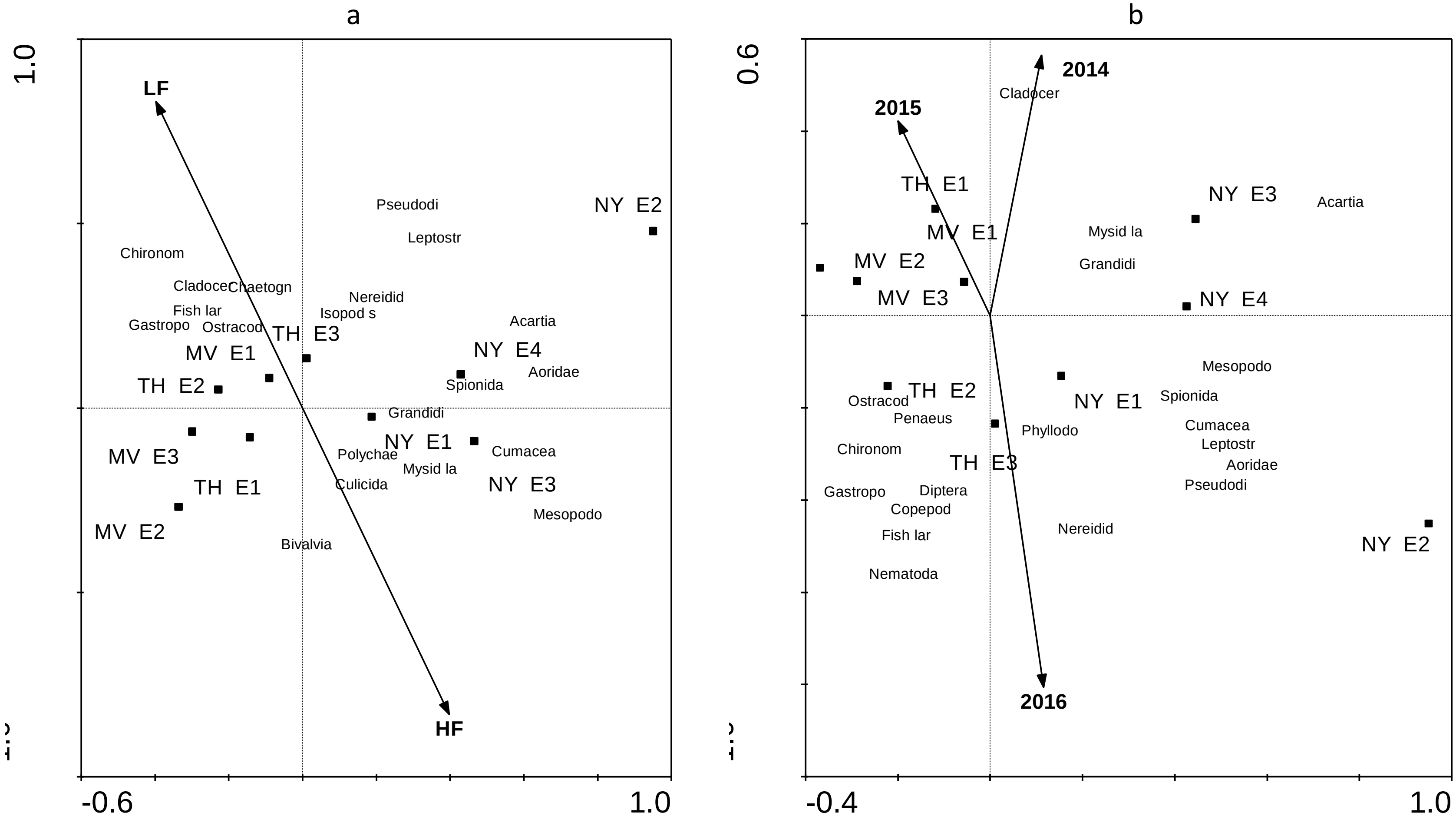

Fig. 3.4: RDA triplots showing the relationship between zooplankton species and (a) flows and (b) years. . (MV-E1-3 = uMvoti Estuary site 1-3; TH-E1-3 = Thukela Estuary sites 1-3, NY- E1-4 = aMatikulu/Nyoni Estuary sites 1-4

3.4 Discussion

3.4.1 Environmental variables

In the current study some physico-chemical parameters between the three estuaries were highly variable as previously found (Chapter 2). Although the three estuaries studied occur in the same geographical area and are geomorphologically similar according to Harrison et al., (2000), the large variability in water quality conditions can be attributed to different land use activities and their varying intensities on these systems as similarly found for macrozoobenthos chapter (Chapter 2). Temperatures between the estuaries were generally comparable with a consistent spatial trend of lower temperatures observed in the lower reaches of the estuaries studied. These spatial trends are attributed to influences from the marine environment which brings in cooler waters from the sea to the lower reaches of these systems. The aMatikulu/Nyoni Estuary had the highest oxygen concentrations while the uMvoti Estuary displayed the lowest. Anthropogenic threats that might be affecting oxygen concentrations in the aMatikulu/Nyoni Estuary are restricted to sedimentation and water quality alteration primarily. Sources of stressors include one sugar mill and associated agricultural activities upstream of the aMatikulu River. The uMvoti River in comparison is highly utilized and affected by multiple upstream sources including industries, waste water treatment works, agricultural activities, water abstraction and urban and peri-urban communities. Anthropogenic threats associated with these users might have resulted in reduction in oxygen levels in this system during the current study. The low to moderate oxygen levels in the Thukela Estuary are also attributed to local catchment land use activities including industries, mining, agriculture, recreation, paper mill and waste water treatment works. High variability in salinity levels between the uMvoti, Thukela and aMatikulu/Nyoni Estuary was observed during the current study. The lower salinity values recorded from the Thukela and uMvoti systems are attributed to these systems being controlled by river discharge more than marine tidal influence. Marine water has very little influence on these systems. The uMvoti Estuary has a limited potential for significant tidal exchange (Badenhorst, 1990). Furthermore, the sea water intrusion in the Thukela Estuary is effective on spring high tide when the river flow is low (from July to September) (Whitfield & Harrison, 2003). Salinities recorded in the three estuaries of this study were within ranges for river dominated estuaries in South Africa (Whitfield, 1992).

Turbidity values varied largely between the three estuaries with the highest turbidity levels recorded from the Thukela Estuary and the lowest from the aMatikulu/Nyoni Estuary. Generally higher turbidity levels were recorded during high flow and this can be explained by associated higher rainfall which results in sediment disturbance increasing the levels of total suspended solids (Froneman, 2002). The Thukela system is facing an increasing pressure on its structure and function due to increasing anthropogenic demand for water resource services (King & Pienaar, 2011). This was evident in the Thukela Estuary with high loads of soft sediments accumulated in the estuary with poor flushing of the river as a result of reduced flow (pers. obs.). This soft sediment accumulation thus increased the turbidity levels in the Thukela Estuary.

Elevated concentrations of DIN during high flow in the Thukela Estuary were consistent with other South African permanently open estuaries (POEs) such as Sundays, Swartkops, Kromme, Kariega and Great Fish estuaries (Allanson & Read, 1995; Grange et al., 2000; Scharler & Baird, 2003). Such higher nutrient concentrations were a result of higher rainfall and high flows leading to increased nutrient input from inland. The aMatikulu/Nyoni Estuary had higher DIN concentration when compared with the uMvoti and Thukela estuaries. Such low concentrations of DIN in the uMvoti and Thukela estuaries can be attributed to the freshwater abstractions in the catchments of uMvoti and Thukela Rivers leading to limited nutrient input (pers. obs.). The exhibition of elevated DIN concentrations in the Thukela Estuary when the river flow is high identifies Thukela River as a viable source of nitrogen to this estuary. Such flow dependent patterns highlight the importance of adequate release policy and adherence to this policy for this heavily utilized estuary. During high flow, dilution effect of DIN and DIP was evident in the aMatikulu/Nyoni and uMvoti estuaries. Dilution of nutrients during high flow has previously been reported in other South African POEs (MacKay, 1993; Scharler and Baird, 2003). Such fluctuations in nutrient concentrations in estuaries largely depends on the quality and quantity of freshwater inflow (Mallin *et al*., 1993). During low flow conditions, seepage from the agricultural land in the upper reaches of estuaries generally becomes the primary source of nitrogen in estuaries (Snow et al., 2000). During these conditions higher nitrate concentrations are measured in the upper reaches (Snow et al., 2000). The uMvoti and aMatikulu/Nyoni estuaries are dominated by agricultural activities upstream. Higher nitrogen concentrations during low flow in these systems are likely to have come from agricultural seepage associated with these activities. Furthermore, higher primary productivity (high chl–a concentrations) was evident in the upper reaches of the uMvoti Estuary during 2016 low flow period highlighting sufficient nutrients for the primary producers in this region of the estuary.

Although some chl–a data for some sampling sessions are absent due to sample volume, chl–a values in the study area were expected to be low because these estuaries are river dominated with little resident time to allow for sufficient primary productivity. Available chl–a data supports this expectation as most chl–a values were generally low in all estuaries although higher values were sometimes recorded during low flow. During the current study, the uMvoti Estuary exhibited higher chl–a concentrations while the Thukela exhibited the lowest with both estuaries experiencing higher chl–a concentrations during low flow. Higher concentrations of nutrients promote primary productivity in POEs (Allanson & Read, 1995; Grange & Allanson, 1995; Froneman, 2002; Perissinotto et al., 2003). Such higher concentrations of chl–a in the Thukela Estuary during low flow were in contradiction with the higher DIN levels on this system which were recorded during high flow. However, in the uMvoti and aMatikulu/Nyoni estuaries, the higher chl–a concentrations during low flow correlated with the higher nutrients concentrations during this period. Similar to the Thukela and aMatikulu/Nyoni estuaries during the current study, higher chl–a concentrations in the lower reaches were also measured in the Berg Estuary (Adams & Bate, 1999). Similar to the uMvoti Estuary during 2015 low, higher chl–a concentrations in the upper reaches were also measured in the Gamtoos, Sundays, Kromme and Swartkops estuaries (Snow et al., 2000; Bate et al., 2002b; Scharler & Baird, 2003). The chl-a concentrations measured in the uMvoti Estuary during the current study were higher than those previously measured in other South African POEs e.g. in the Sundays, Kromme, Kariega, Great Fish River and Gamtoos estuaries (Grange et al., 2000; Bate et al., 2002a, Scharler & Baird, 2003). Phytoplankton bloom is defined as chl–a concentration greater than 20 µg l^{-1} (Adams & Bate, 1999). Chlorophyll a concentrations measured in the uMvoti Estuary during 2015 low flow were greater than 20 µg l^{-1} and this depicts that this system experienced a phytoplankton bloom during this period. Concentrations of chl–a in the Thukela and aMatikulu/Nyoni estuaries during the current study were higher than those previously recorded in other South African POEs such as Kromme, Kariega and Mlalazi estuaries, however these concentrations were lower than those previously recorded in Sundays, Great Fish River and Gamtoos estuaries (Grange et al., 2000; Snow et al., 2000; Bate et al., 2002b, Scharler and Baird, 2003; Vezi, 2013). The current study suggests that the three estuaries studied have little potential for high primary production.

Although the uMvoti and Thukela estuaries are affected by relatively similar land use activities and although these two systems are impacted by water abstraction activities upstream, the lower phytoplankton biomass (chl–a) in the Thukela Estuary is likely to have been a result of higher turbidity levels in this system (pers. obs.). Turbid waters of the Thukela Estuary limit light penetration due to reduced clarity and this may prohibit primary productivity. The Thukela Estuary is classified as River Mouth according to Whitfield (2000) and such systems have short residence time which limits primary production duration and phytoplankton accumulation. This might be the other reason for the reduced chl–a concentrations in the Thukela when compared with the other estuaries studied. The significance of retention time on phytoplankton biomass have also been reported in other studies (Hilmer & Bate, 1990; Cromar & Fallowfield, 1997). Phosphate may be limiting to phytoplankton production in the Thukela Estuary, owing to its low concentrations during the current study. Reduced pelagic chl–a values in the three estuaries during high flow of the current study suggests the impact of estuarine flushing which might have washed phytoplankton to the adjacent sea. Again, strong river flow might have reduced water residence time which is essential for nutrient utilization by phytoplankton.

3.4.2 Zooplankton

Information on zooplankton communities in the uMvoti, Thukela and aMatikulu/Nyoni estuaries is sparse. Zooplankton abundances in the uMvoti and Thukela estuaries were higher during low flow as opposed to the aMatikulu/Nyoni Estuary which exhibited higher abundance during high flow. Freshwater flow is one of the main parameters controlling zooplankton seasonal variations in estuaries (Chicharo et al., 2006). Higher zooplankton abundance in aMatikulu/Nyoni Estuary during high flow may be attributed to high nutrients and sediments carried into the estuary and thus promoting primary productivity, which in turn favors high zooplankton abundance. Low abundance of zooplankton during high flow in the uMvoti and Thukela estuaries could be a result of outflow of estuarine water washing away the zooplankton into the adjacent sea. Throughout the study, calanoid copepods particularly *A. natalensis* and *P. hessei* remained the most dominant species in all the three estuaries studied. The next most abundant taxa during the present study were Chironomidae, Nematoda and *M. africanus* in the uMvoti, Thukela and aMatikulu/Nyoni estuaries respectively. Dominance of copepods in the three estuaries studied is a typical phenomenon for estuaries of South Africa (Wooldridge, 1999; Jerling, 2005). The uMvoti and Thukela estuaries had salinities of less than 4 for most of the study period. This explained the relative dominance of freshwater taxa on these systems. As previously reported, typical estuarine species dominate mesohaline waters while freshwater organisms dominate oligohaline (salinity < 4) waters (Wooldridge & Bailey, 1982; Wooldridge, 1999). Zooplankton abundance in the aMatikulu/Nyoni Estuary displayed a different spatial distribution pattern to that of other South African POEs. Abundance in the aMatikulu/Nyoni Estuary was lower in the upper reaches increasing down the salinity gradient. Zooplankton abundance was highest in the upper reaches decreasing down the estuary in the Breede, Great Berg, Goukou, Heuningnes, Kromme and Olifants estuaries (Wooldridge & Callahan, 2000; Montoya-Maya & Strydom, 2009). The mean zooplankton abundance recorded in the aMatikulu/Nyoni Estuary during the present study (15087 ±10865 ind. m^{-3}) was higher than that previously recorded in other South African estuaries e.g. in the Breede (mean = 4049 ind. m^{-3}), Heuningnes (mean = 3877 ind. m^{-3}), Goukou (mean = 6175 ind. m^{-3}), Olifants (6269 ind. m^{-3}) and Great Berg estuaries (mean = 6841 ind. m^{-3}) (Montoya-Maya & Strydom, 2009). However, the zooplankton abundance recorded in the aMatikulu/Nyoni Estuary was 2-fold lower than that previously recorded in the Mlalazi Estuary (Vezi, 2013). Mean zooplankton abundances recorded in both uMvoti and Thukela estuaries were lower than those previously recorded in other South African estuaries e.g. Goukou, Breede, Heuningnes, Great Berg, Olifants and Mlalazi estuaries (Montoya-Maya & Strydom, 2009, Vezi, 2013). Zooplankton abundance was generally higher in the lower reaches of the uMvoti and Thukela estuaries during the current study.

Similar to the aMatikulu/Nyoni Estuary, water temperature had an effect in structuring zooplankton community in other South African POEs such as Sundays, Gamtoos and Kromme estuaries (Wooldridge & Bailey, 1982; Jerling & Wooldridge, 1991; Schumann & Pearce, 1997). As observed in the aMatikulu/Nyoni Estuary during the current study, dissolved oxygen has been reported to control zooplankton community structure in the estuaries of Bilbao and Urdaibai (Albaina et al., 2009). Salinity was identified as the key environmental variable in structuring the plankton communities in South African estuaries (Collins & Williams 1982; Wooldridge, 1999). Similarly, salinity was one of environmental parameters responsible for structuring zooplankton community in the Estuary during the current study. Turbidity and pH were determinants in structuring zooplankton communities in the uMvoti and Thukela estuaries. Zooplankton abundance and community structures have been reported to be controlled by pH, temperature, salinity, dissolved oxygen and turbidity in other parts of the world (Laprise & Dodson, 1994; Pandey & Verma, 2004; Tackx et al., 2004; Uriarte & Villate, 2004; David et al., 2005; Albaina et al., 2009; Mialet et al., 2011; Almeida et al., 2012; Farhadian & Pouladi, 2014).

Available chl-a data from the current study displayed relationship with the zooplankton abundance in the uMvoti and Thukela estuaries. Zooplankton abundance increased with phytoplankton biomass (chl-a) in the lower reaches of the Thukela Estuary during 2015 low flow. A similar pattern was observed in the lower reaches of the uMvoti Estuary during the same sampling period. In both uMvoti and Thukela estuaries zooplankton abundance decreased with decreasing chl-a concentrations during high flow. Such relationship between the zooplankton abundance and chl-a concentrations suggests the potential effect of phytoplankton availability on the zooplankton abundance. Such pattern was also reported in the Great Fish, Sundays and Kariega estuaries (Wooldridge & Bailey, 1982; Jerling & Wooldridge, 1991; Grange et al., 2000) and in other parts of the world e.g. Scheldt Estuary in Belgium (Mialet *et al*., 2011) and Golden Horn Estuary in Turkey (Dorak & Albay, 2016). No clear relationship was observed between zooplankton abundance and chl-a concentrations in the aMatikulu Estuary. This might suggest that chl-a may be less important in controlling zooplankton abundance compared with water quality in this Estuary. A similar trend was reported in the Scheldt Estuary (Mialet et al., 2011). Turbidity levels in the Thukela Estuary were higher than those recorded in the aMatikulu Estuary during the current study. Such high turbidity levels in the Thukela Estuary might have resulted in the lower zooplankton abundance in this system when compared with the aMatikulu Estuary. High turbidity levels may affect zooplankton survival by restricting selective feeding and fecundity of these organisms (Sellner & Bundy, 1987; Gasparini & Castel, 1999).

Most estuaries in South Africa experience reduced zooplankton abundance during low flow compared with high flow (Wooldridge, 1999). Higher zooplankton abundance in the aMatikulu/Nyoni Estuary during high flow was consistent with this pattern. This pattern was similarly observed in other South African permanently open estuaries (Montoya-Maya & Strydom, 2009). However, the pattern observed in the uMvoti and Thukela Estuary was in contrast to this as higher abundances were recorded during the low flow in these estuaries in the current study. The lower zooplankton abundance in these two estuaries during high flow might be attributed to the washing of zooplankton with estuarine water to the adjacent marine environment. The similar seasonal pattern of zooplankton abundance between these two estuaries was expected from the close geographical location of both systems accompanied by similar function of these systems. Although the uMvoti had higher chl–a concentrations when compared with the Thukela and aMatikulu/Nyoni estuaries, the lower zooplankton abundances in this system are likely to be attributed to the short residence time in this system together with low oxygen concentrations recorded in this Estuary which reflected the higher degree of pollution in this system. This higher degree of pollution in the uMvoti Estuary was reflected in the zooplankton community by showing lower abundance and lower number of taxa when compared with Thukela and aMatikulu estuaries. Copepods which were the main dominant group in this system are known to have low tolerance to reduced oxygen levels (Roman *et al*., 2009), hence their low abundances. Compared with the Thukela Estuary, higher chl-a concentrations recorded in the uMvoti Estuary might be related to nutrient enrichment from anthropogenic sources as higher nutrient concentrations were recorded in the uMvoti when compared with the Thukela Estuary. Higher zooplankton abundance in the aMatikulu/Nyoni Estuary was likely to be attributed to the relatively higher phytoplankton biomass, higher nutrient levels, higher oxygen concentrations as well as sufficient residence time for both phytoplankton to utilize nutrients and zooplankton to utilize phytoplankton efficiently.

Pseudodiaptomus hessei was recorded in high abundance after rains and floods in the Sundays and Swartkops (Wooldridge & Bailey, 1982; Wooldridge & Melville-Smith ,1978). The dominance of *P. hessei* during low flow in the uMvoti and Thukela estuaries during the current study was in contradiction of this pattern. The dominance of this species during low flow was not surprising since it has been recorded in salinities ranging between 0 and 80 (Grindley, 1981). Furthermore, salinities were relatively low during low flow in both uMvoti and Thukela estuaries allowing *P. hessei* to thrive on these systems during this period. Numbers of *A. natalensis* in the aMatikulu/Nyoni Estuary were very low when compared with *P. hessei* and *M. africanus. Acartia natalensis* remains permanently in the water column (Kibirige & Perissinotto, 2003), this may get these organisms washed to the sea during high flow. Salinity values recorded in the aMatikulu/Nyoni Estuary ranged from 1.4 to 53.9 and such variation in salinity may explain the low numbers of *A. natalensis* as these organisms are vulnerable to fluctuations in salinity as supported by Jerling & Cyrus (1999). The mysid *M. africanus* displays an opportunistic behavioral response to low salinity as a result of freshwater inflow (Owen & Forbes, 1997; Kibirige & Perissinotto 2003). Similarly, this species was recorded in high numbers during high flow in the aMatikulu/Nyoni Estuary during the current study. Mysids, particularly *Mesopodopsis* species was reported to be positively correlated to salinity in Gironde Estuary (France) (David et al., 2005). Similarly, this species is known to be controlled by salinity in other North European estuaries (Mees et al., 1993; Azeiteiro & Marques, 1999; Mouny et al., 2000).

3.5 Conclusions

The spatial variability in the zooplankton distribution and abundance can be explained by the horizontal salinity gradient in the three estuaries studied. The results of the current study showed that spatial and temporal variation in the water physico-chemical parameters of the three estuaries studied have significant effect on the structure and abundance of zooplankton assemblages. This may potentially affect the biodiversity and functioning of these estuaries. Significant variation in temporal zooplankton distribution, species composition and abundance may be attributed to river inflows and changing environmental parameters. The high turbidity characteristic of the Thukela Estuary was among the important factors in determining the copepod temporal variability as this results in alterations in copepod's selective feeding. In addition, high turbidity is responsible for very low primary production which is a measure of food availability for zooplankton. Our study showed that zooplankton responds to the seasonal and spatial variability in environmental variables. This study also showed that the impacts of land use activities taking place in the catchments of the three estuaries are reflected in the water quality and zooplankton communities of these estuarine systems. This was apparent in the uMvoti and Thukela estuaries with altered water quality accompanied with lower zooplankton abundance when compared with the aMatikulu/Nyoni Estuary. Threats to the aMatikulu/Nyoni Estuary are generally minimal with little anthropogenic impacts on the estuary currently which include only one sugar mill and associated agricultural activities upstream of the aMatikulu River. The uMvoti Estuary, however, is largely affected by pollution from different anthropogenic land use sources further upstream including effluent input of treated sewage, sugar and paper mill effluents from Gledhow Sugar Mill and Sappi Stanger Pulp and Paper Mill, agricultural irrigation and several domestic uses by urban and informal settlements. The anthropogenic land use activities associated with the Thukela catchment include water abstraction for industrial and domestic use, industries, agriculture, mining, recreation, waste water treatment works, paper mill, and road and rail networks. We conclude that the environmental variability and seasonality in river inflow are the important factors influencing zooplankton distribution, species richness and abundance in the uMvoti, Thukela and aMatikulu estuaries. Changes in the environmental factors (e.g. oxygen, turbidity and chl-a) as a result of land use need to be monitored and the land use activities need to be properly managed to reduce their impacts on the estuarine systems. Changes in zooplankton communities and diversities associated with altered environmental factors need to be closely monitored. Should land use management be implemented to reduce impacts, then water quality might improve and this might result in zooplankton responding to these changes and such response need to be monitored.

Findings of the current study suggest that the three estuaries studied require appropriate management to ensure adequate freshwater supply so as to maintain their good ecological structure and function. The occurrence of Chironomidae (a pollution tolerant insect) in higher numbers in the uMvoti Estuary indicated that this system is in a poor ecological state with altered water quality. Although the number of taxa in the Thukela Estuary was similar to the aMatikulu/Nyoni Estuary, lower zooplankton abundances in the Thukela Estuary coupled with relatively altered water quality were indicative of relatively poor ecosystem wellbeing.

In the current study, some physico-chemical parameters between the three estuaries were highly variable and identified as issue of particular concern. The anthropogenic land use activities that are associated with the three estuaries studied differed. Although the three estuaries studied occur in the same geographical area and are similar in function according to Harrison et al., (2000), the large variability in water quality conditions can be attributed to different land use activities and their varying intensities on their catchments. Variability in zooplankton community structures in these three estuaries are also likely to be attributed to the alterations in water quality in particular and habitat conditions. Most water quality parameters were in more acceptable levels in the aMatikulu Estuary compared with the uMvoti and Thukela estuaries and this is supported by relatively higher zooplankton species richness and abundance in this system. Land use activities associated with the uMvoti and Thukela estuaries are likely to have resulted in the reduction in oxygen levels in these systems observed during the current study. Large variation in turbidity levels between the three estuaries were also observed with the highest turbidity values recorded in the Thukela Estuary and the lowest recorded in the aMatikulu/Nyoni Estuary. This is evident in the Thukela system with high load of sediments accumulated in the estuary. Such sediment accumulation is attributed to the poor flushing of the Thukela River as a result of reduced flow. This sediment accumulation thus increased the turbidity levels in the Thukela Estuary. The Thukela Estuary displayed a relatively higher number of zooplankton taxa during the current study and it has been previously acknowledged for its nursery function. Although there is an increased pressure on the structure and function of the Thukela system as a result of increasing demand for water related ecosystem services, the lower portion of the Thukela River and its associated estuary are still characterized as an ecologically important region that provides a range of services and thus require proper management.

The aMatikulu Estuary remains in a relatively good ecological state when compared with the uMvoti and Thukela estuaries in terms of abundances and species richness and this system lies within a nature reserve. However, new changes in water quality together with an increase in sediment load suggests that new stressors are acting on its catchment. This estuarine system needs to be conserved and monitored to prevent stressors from affecting the community structures and functioning of this system. The uMvoti Estuary is heavily impacted with a shift in communities, loss of biodiversity and nursery function. The uMvoti Estuary catchment is heavily affected by the anthropogenic water resource use activities as described above. The impacts of these activities is reflected in the water quality and habitats of this system which in turn is reflected in the zooplankton communities comprising low diversities and abundances. The uMvoti riverine system has been modified completely, with nearly total loss of natural habitat and biota and the destruction of many basic ecosystem functions (Tharme, 1996). As a result, the uMvoti Estuary is regarded as a degraded system which functions differently from the way it did in its former pristine state (MacKay and Cyrus, 2000). Since 1964, water quality of the uMvoti Estuary was described as grossly polluted (Begg, 1978). This may explain the dominance of the insects Chironomidae in the uMvoti Estuary. Although the uMvoti system is heavily threatened and polluted, management measures are needed so as to improve its water quality and biodiversity and also to regain its nursery function.

River dominated estuaries have short residence time. These estuaries are usually small, which is what was observed in this study (< 3 km^2) and they allow most matter to pass through the mouth into the sea before it is deposited. This was confirmed in our case study by generally low chl-a values and zooplankton abundance when compared with other permanently open estuaries. Lower abundances in these systems are associated with estuary flushing which washes zooplankton organisms and phytoplankton to the adjacent marine environment. Although the natural factors such as flow seasonality, environmental variables and estuary morphology can determine zooplankton community structures (Laprise and Dodson, 1994; Pandey and Verma, 2004; Tackx et al., 2004; Uriarte and Villate, 2004; David et al., 2005; Albaina et al., 2009; Mialet et al., 2011; Almeida et al., 2012; Farhadian and Pouladi, 2014), in our study the river dominated estuaries also responded to changes in anthropogenic land use activities and their water quality, quantity and habitat altering stressors. In this study we showed that river dominated estuaries can also be vulnerable to sedimentation as observed in the Thukela Estuary. Flow reductions exacerbate these impacts. The response of zooplankton to other altered environmental variables including oxygen also showed that the river dominated estuaries are vulnerable to anthropogenic land use activities that need to be managed to achieve a suitable balance between the use and protection of ecosystems which conforms to best Integrated Water Resource Management approaches (DWA, 2013).

If these three estuaries together with the land use activities in their catchments are not managed, they will continue to deteriorate. This may lead to a complete loss of nursery function as observed in uMvoti Estuary, loss of biodiversity and severely altered water quality. We recommend that remediation and mitigation focus should be directed to multiple sources of impacts and not just one source of interest/concern as this will have higher efficiency on reducing impacts to the ecological functining of the uMvoti, Thukela and aMatikulu/Nyoni estuaries. Estuary Management Plans are urgently needed for these three estuaries so as to establish protection, conservation and management measures needed to minimise impacts, with the Thukela Estuary being the priority followed by the aMatikulu and then uMvoti Estuary. Restoration of riparian vegetation of the estuaries studied can aid in improving water quality and aquatic habitats of these systems. Development of riparian buffers may be another important strategy to reduce sediment loading and erosion into these impacted estuaries.

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CHAPTER 4

Application of the Relative Risk Model for evaluation of ecological risk in selected river dominated estuaries in KwaZulu-Natal, South Africa.

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Abstract

Effective environmental management and restoration of impacted estuaries in South Africa necessitates a holistic understanding of the contribution of various stressor-related impacts throughout the catchment. Ecological risk assessment for aquatic ecosystems is important to water resource management. In this study, we describe results of a preliminary assessment that was conducted to evaluate the relative risks of multiple anthropogenic stressors currently acting within the catchments of uMvoti, Thukela and aMatikulu/Nyoni estuaries using Bayesian Network Relative Risk Model (BN-RRM) framework. Four socio-ecological endpoints selected for the current study included biodiversity habitat, safe environment, fishery and productivity. We constructed a conceptual model which depicted potential and effect pathways from the source, to the stressor, to the habitat and to the endpoint. We also developed five scenarios (including historical and future scenarios) to predict the potential risk distributions in different proposed scenarios. Results revealed that productivity was the endpoint at the lower risk in all the estuaries and all scenarios except for scenario 5. Results also showed that scenario 3 which is a scenario before major resource development had the lowest risk scores for all the endpoints. Scenario 4 (year 2025 if no laws and management measures are implemented) had the highest risk scores for all the endpoints. Overall endpoints generally displayed low to medium risk throughout all scenarios (except scenario 3) and different flows. All endpoints generally displayed zero risk in scenario 3. All endpoints were at a highest risk in the uMvoti Estuary followed by aMatikulu/Nyoni and then Thukela Estuary. Results highlighted that in the uMvoti and Thukela estuaries, people are at a higher risk when compared with the ecological components of these systems as social endpoints displayed higher risk scores than the ecological endpoints, however the opposite was observed in the aMatikulu/Nyoni Estuary. This study provided the foundation for evaluating the risks of multiple stressors in the catchments of these estuaries to a variety of endpoints. Management options and research should focus on collecting necessary data and information to refine the developed RRM. By establishing such framework, we believe that stakeholders within the catchments of these systems together with government organizations will be able to make more informed and risk-based management decisions pertaining restoration and rehabilitation options for these three estuaries.

Keywords: Ecological risk assessment, estuaries, multiple stressors

4.1 Introduction

Risk assessment is defined as the process of assigning magnitudes and probabilities to the adverse effects of anthropogenic activities or natural disasters (Suter, 1993). Ecological Risk Assessment (EcoRA) is defined as an organised approach that describes, explains and organises scientific facts, laws and relationships, so as to provide a sound basis to develop adequate protection measures for the environment (U.S.E.P.A, 2008). Ecological risk assessment is an essential tool for environmental management and has been widely applied in environmental decision making (Nash et al., 2005). The purpose of EcoRA is to provide information of a given stressor profile regarding its impacts on an ecosystem so that pollution and other eco-environmental damages can be minimised (Suter, 1993). A regional-scale ecological risk assessment can be defined as a summary of the complex interactions and effects of chemical and non-chemical stressors on the ecological endpoints (Wiegers et al., 1998). Regional ecological risk assessment looks at a spatial scale that comprises of multiple habitats with multiple sources of multiple stressors affecting multiple endpoints with the characteristics of the landscape affecting the risk estimate (Landis, 2005). At a regional scale, there are usually multiple sources for a single stressor (Liu et al., 2010). A regional scale risk assessment using the Relative Risk Model (RRM) is a form of EcoRA that is performed on a spatial scale where considerations of multiple sources of multiple stressors affecting multiple endpoints are allowed (Landis, 2005). In addition, allowance for the landscape characteristics that may affect the risk estimate is made possible. Similar to traditional EcoRA approaches, RRM may be carried out to assess the risk posed by one stressor of concern to one endpoint. However, at a regional scale, multiple stressors acting on a range of ecological endpoints can be considered within a RRM framework (Landis and Wiegers, 1997). The relative risk model was established so as to integrate the impacts as a result of variety of stressors at a regional scale (Landis and Wiegers, 1997; Wiegers et al., 1998). The RRM has also been applied successfully in other parts of the world including Brazil (Moraes et al., 2002), Australia (Walker et al., 2001), USA (Hayes and Landis, 2004; Obery and Landis, 2002; Wiegers et al., 1998) and China (Liu et al., 2010).

Estuaries have high socio-economic value as they provide goods and services such as water, sand, pollution control, recreation, harbours and fish for people. Estuarine systems can serve as filters because of their capacity to trap excess nutrients coming from inland (Telesh and Khlebovich, 2010; Turpie et al., 2002). These systems are also used by industries for disposal, processing and transportation of waste (Onojake et al., 2011). Estuaries are among the most threatened systems by anthropogenic impacts and this may alter their ecological functioning which includes their ability to act as nursey grounds (Nicolas et al., 2007). Many estuarine systems along the north coast of KwaZulu-Natal (KZN) South Africa, are threatened by the poor water quality, reduced flows and habitat alterations originating from land use activities (King & Pienaar 2011).

The uMvoti Estuary (KZN) has been rated severely degraded in terms of sedimentology and this condition deteriorates with time due to high sediment loads during flooding conditions (Badenhorst, 1990). The ecological functioning of the Thukela Estuary (KZN), the lower Thukela River and near shore marine and Thukela Banks ecosystems have deteriorated over the last few decades. The impacts and deterioration are linked to changes in land use within the catchment and alteration in volume, timing and duration of flows by abstraction to meet the demand of users (DWAF, 2004). The aMatikulu/Nyoni Estuary (KZN) is considered to be in a good condition although siltation from the catchment is of concern (Whitfield, 2000). No previous study has been conducted to assess the risk of stressors to the endpoints of these three estuaries using Bayesian network Relative Risk Model (BN-RRM) approach. This study aimed at applying the regional ecological risk assessment framework and estimating risk from stressors to selected socio-ecological endpoints. We established three objectives for this study. The first objective was to develop a RRM to estimate the relative contribution of risk from stressors to selected socio-ecological endpoints in the uMvoti, Thukela and aMatikulu/Nyoni estuaries, South Africa. The second objective was to determine which regions/ estuaries and endpoints were at highest risk from anthropogenic activities. The third objective was to incorporate historical (before major industrial development) and future (including a scenario with implementation of management/mitigation measures and a scenario without implementation of management/mitigation measures) scenarios into the model and estimate the relative contribution of risk to selected socio-ecological endpoints.

4.2 Methods

The first step was to give description of study sites and risk regions. Next, we gave a detailed description of the relative risk model, including the initial construction of the model framework and model parameterization.

4.2.1 Study area and sub-regions

The present study was conducted in the three river dominated estuaries in the North Coast of KwaZulu-Natal (KZN), South Africa. The uMvoti, Thukela and aMatikulu/Nyoni estuaries (Chapter 2) lie in the same geographical area and are similar in geomorphology (Harrison et al., 2000) (Fig. 4.1). The uMvoti and Thukela estuaries are classified as river mouth while the aMatikulu/Nyoni Estuary is classified as a permanently open estuary (Whitfield, 2000). In the current study, each estuary was considered as a risk region. The uMvoti Estuary (29^{o}23' S, 31^{o}20' E) (Chapter 2, Fig. 2.1) is a subtropical river mouth which occupies a surface area of 0.2 km^2. The uMvoti River catchment is subject to agricultural activities which include commercial forestry, sugar cane farming, commercial dry land agriculture and subsistence farming. The Thukela Estuary (29^{o}13' S, 31^{o}29' E) (Chapter 2, Fig 2.1) is a subtropical river mouth occupying a surface area of approximately 0.6 km^2 (Begg, 1978; Whitfield, 2000). The ecological importance of the Thukela Estuary includes transportation of sediment into the local marine ecosystem which is linked to the ecological functioning of a biodiversity hot-spot in a near shore marine environment and the commercially important Thukela Banks Fisheries (DWAF, 2004). The Thukela Estuary act as a conduit for many anadromous species thriving in the upper reaches and this system can serve as a habitat for some estuarine resident, marine migrants and freshwater fish species (DWAF, 2004). Anthropogenic activities associated with ecosystem services use in the lower Thukela River and estuary include water abstraction for domestic use, industries, agriculture, mining, recreation, waste water treatment and road and rail networks. The aMatikulu/Nyoni Estuary (36^{o}06'36"S, 31^{o}37'09"E) (Chapter 2, Fig. 2.1) is a subtropical permanently open estuary which covers a surface area of 900 km^2 (Begg, 1978; Whitfield, 2000). The lower reaches of the aMatikulu/Nyoni Estuary are disturbed by agricultural activities but the fauna is generally in a good condition (Harrison et al. 2000). This estuarine system forms part of the aMatikulu Nature Reserve, managed by Ezemvelo KwaZulu-Natal Wildlife (EKZNW). Agricultural activities taking place in the catchment include sugarcane and subsistence farming with some commercial forestry. There is one sugar mill upstream of the aMatikulu/Nyoni Estuary.

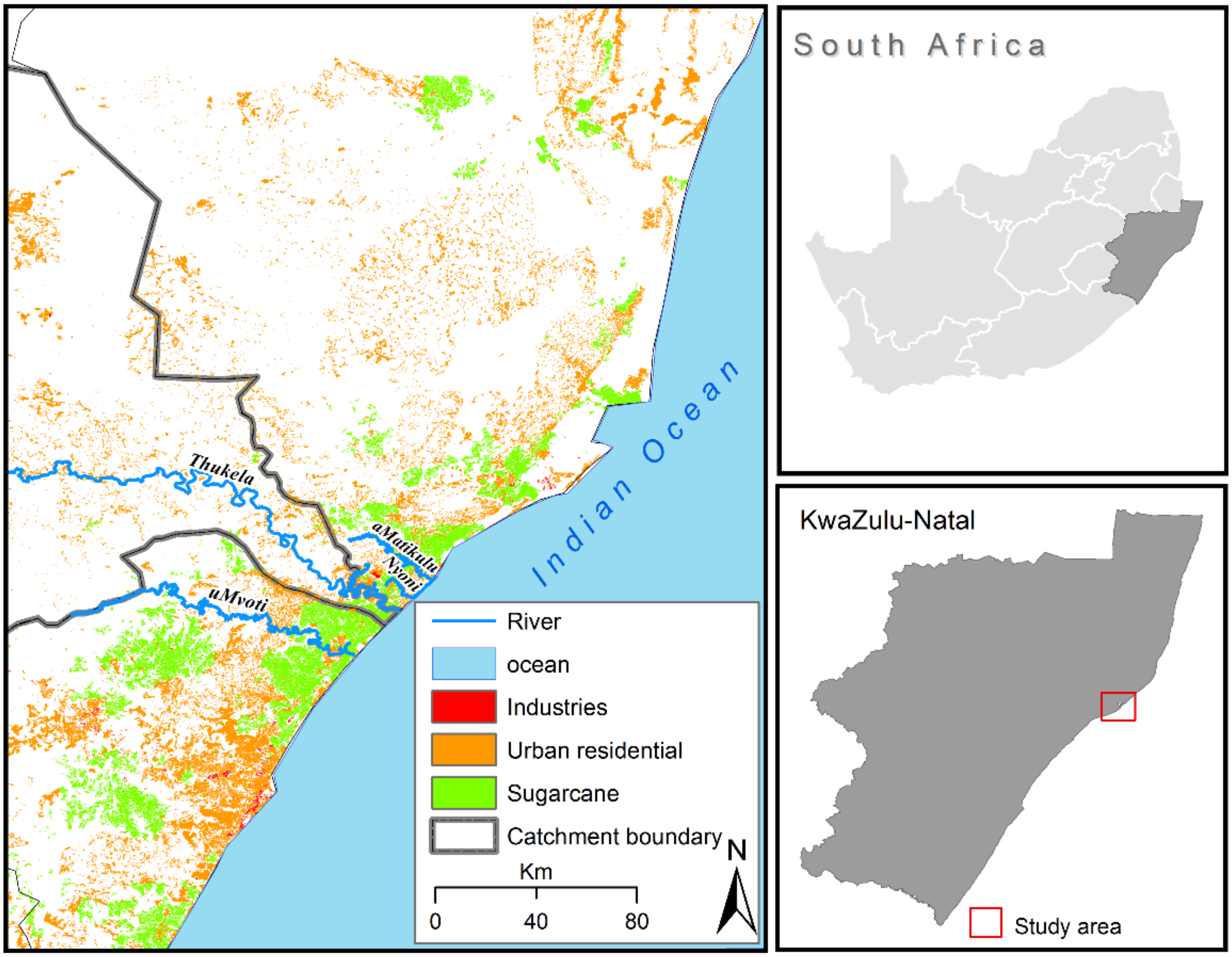

Fig. 4.1: Location of uMvoti, Thukela and aMatikulu estuaries with associated major land use in their respective catchments.

4.2.2 Assessment endpoints

The three estuaries considered in the study lie in the same geographic region, however, they are impacted by anthropogenic activities of different nature and intensity. It is acknowledged that the way in which these systems respond to anthropogenic impacts is poorly understood. The ecologically important species that can display significance of regional ecological risk assessment in a region is usually selected as an ecological endpoint (Liu et al., 2010). The current study focused on the aquatic environment of the three estuaries. In this case study endpoint is the management objective that is definitive, something that we care about and we know where it is. The assessment endpoints that were selected for the present study were biodiversity habitat, safe environment, fishery and productivity. These endpoints were chosen because they consider habitat for zooplankton, macrozoobenthos and fish, in addition these endpoints have both ecological and social value. Sources of stressors could release stressors which may cause effect to the endpoint and endpoint could be vulnerable to the stressor in these particular habitats. In this case study we selected biodiversity habitat as a proxy for biodiversity based on the premise that if we protect the habitat we protect the biodiversity. The focus of this endpoint was on the habitats for biodiversity. Pelagic and benthic habitats were of the main concern on the ecological aspect of this case study. Biodiversity habitat can be affected and degraded by land use activities including pollution, nutrient enrichment, erosion and sedimentation. Safe environment as an endpoint is important for both humans and aquatic organisms. If parameters like water quality and sediment load are altered and not managed, the safety of the environment will be compromised. We selected safe environment as a proxy of safety for aquatic fauna and flora as well as human population to cover socio-ecological aspect in the case study. This takes a measure of water quality which affects both aquatic organisms as well as human populations (through swimming and fishing). Safe environment endpoint also takes a measure of depth which on a social aspect measures the safety of human population and this is controlled by sedimentation and mouth state. Productivity of an estuary as an endpoint is essential for both primary and secondary production and if water quality and flows of the estuary are modified, productivity is likely to be altered. Water quality parameters like nutrients and turbidity (determinant of sunlight penetration) play an important role in primary productivity of an estuary. Flows together with mouth state can also determine the residence time of an estuary which in turn can determine the primary productivity of an estuary. In this case study we selected productivity as an endpoint as some of the studied estuaries are affected by sedimentation and high turbidity levels coupled with altered water quality and thus it needed management attention. Alteration of water quality and nutrient levels will affect primary productivity which in turn will affect secondary production of the system. Fishery endpoint was selected to cover socio-ecological aspect in the case study. From the conceptual model this endpoint is determined by productivity (both primary and secondary) which serves as a proxy for food availability for fish. On the social aspect fishery was considered as a measure of fish availability (catches) for human population consumption. If we protect the environment we are protecting habitats for productivity and thus protecting habitats for fish. The selected endpoints are concerned about the biodiversity and good ecological state of the estuaries and social needs of human population.

4.2.3 Sources and stressors

A wide variety of anthropogenic factors have exerted pressures in the uMvoti (Begg, 1978; DWA, 2004; MacKay et al., 2000; Tharme, 1996) Thukela (DWAF, 2004b; DWAF, 2004c; Ferreira et al., 2008; IWR, 2003; IWR, 2004; O'Brien and Venter, 2012; O'Brien et al., 2009; Pienaar, 2005; Stryftombolas, 2008) and aMatikulu/Nyoni estuaries (Harrison et al. 2000). The anthropogenic impacts that are acting on these three estuaries, although exerting different intensities, include agriculture, industries, abstraction, exotic plants, waste water treatment works, road and railway bridge, urban areas, settlements and recreational activities. If the threat/chemicals produced by the source have a potential of reaching a habitat then a stressor may result. The possible stressors that may result include water quality alteration, water quantity alteration, habitat alteration, disturbance to wildlife and erosion/siltation.

4.2.4 Habitats

Habitats, which are directly related to aquatic environment of the estuary and for which there are spatial and temporal data available were selected, i.e. benthic as well as pelagic habitat. Organisms which inhabit benthic and pelagic habitats in this study are macrozoobenthos and zooplankton respectively, and these are the receptors in these aquatic habitats.

4.2.5 Relative risk model development and risk analysis

Filters and interactions of sources, stressors, habitats, receptors and endpoints were defined based on historical data and the data collected during bio-physical assessment of these three estuaries during the study period. Conceptual model was developed to illustrate interactions among the components (Fig. 4.2c). Furthermore, based on the conceptual model, the representation (Bayesian network) of the predicted relationships among the stressor, the exposure scenarios and assessment endpoint responses was then developed using Netica by Norsys Software Corp (Fig. 4.2d). The relative risk model is based on a ranking scheme that describes the impacts of each stressor on different habitats and their related ecological receptor as well as endpoints. Figure 4.2b depicts this process as adapted from Landis (2005). For the implementation of RRM, each habitat and stressor category was ranked based on relative magnitude throughout the estuary. Information for rank and scores (of which some was obtained from the results in the previous two chapters), as well as rank justification is given in supplementary data (S1 Table 4.1). Ranking of the stressors was based on both quantitative and qualitative information which was used to determine the relative magnitude of each stressor in each estuary. Stressors were assigned to each magnitude category and score as follows: zero, low, moderate and high and these were assigned scores as 25, 50, 75 and 100 respectively. Zero (25) depicted no risk posed by the stressor to the socio-ecological endpoint, low (50) depicted very little/ negligible magnitude of stress to the endpoint, moderate (75) depicted moderate risk of a stressor to the endpoint and high (100) depicted high risk posed by the stressor to the endpoint. The score of zero (25) represented the condition comparable with pristine pre-anthropogenic activities conditions with no possibility of risk to the endpoint wellbeing. This score is comparable with the South African Department of Water and Sanitation (DWS) classification category "A". The score of low (50) represented a suitable sustainable management condition where changes from reference conditions (pre anthropogenic conditions) are evident but do not pose a significant change to biological communities. This score is comparable with the DWS classification category "B and C". The score of moderate (75) represented moderate change without significant loss of biodiversity and processes. This state is proposed to be sustainable but represents the threshold of potential concern where management action is required to avoid high risk conditions. The moderate score is comparable with the DWS classification category "D". The score of high (100) represented unsustainable state with significant changes and it proposed potential irreversible loss of diversity and processes. This score is comparable with the DWS classification category "E-F". Available quantitative data were used to rank system variables, nutrients, river discharge, organic content and sediment grain size. These were based on previous published and unpublished biomonitoring reports (1999-2013) and unpublished theses. Discharge data (flows) were obtained from the Department of Water and Sanitation, South Africa. Some expert judgment was also used to rank some stressors in the study area. Stressors that could not be practically measured during the present study were ranked as a measure of percentage land use cover that may be contributing sources of stressors e.g. toxicity and sediment load. For sediment load we considered man made modifications to the catchment which are likely to cause erosion problems, increased sediment levels as well as large silt loads. For toxicity we considered land cover of toxicant pollution sources in the catchment including agriculture, industries and urban centers. After characterization and prioritization of stressors using RRM, the resource managers or stakeholders will thus be able to focus on identifying the specific sources contributing to a specific stressor in a given estuary (Iannuzzi et al., 2009). It is acknowledged that directing the model to the specific sources rather than the stressors results in greater uncertainty to the model results (Iannuzzi et al., 2009). An important limitation in RRM of the current study was the natural variability in the response of receptors to changes in habitat. The RRM as applied to the uMvoti, Thukela and aMatikulu/Nyoni estuaries regional risk assessment was established on the following assumptions:

- The potential for exposure (i.e. contact with a stressor) can be determined by spatial overlap of a stressor and habitat;
- The possible differences in exposure can be determined by relative rankings of stressor magnitude in each estuary;
- All habitats have equal importance to structure and function of the ecosystem;
- The density of available ecological receptors is linked to the amount and type of habitat;
- The intensity of effects between risk regions (different estuaries) can be characterized by the relative exposures as well as abundance and characteristics of available receptors; and
- Risks can be ranked relative to each other and inferences can be established pertaining the relative magnitude of risk within each estuary.

The above mentioned simplifying assumptions were used to support the initial application of the RRM to the uMvoti, Thukela and aMatikulu/Nyoni estuaries and are consistent with the assumptions used in other RRM applications e.g. (Iannuzzi et al., 2009; Landis and Wiegers, 2005).

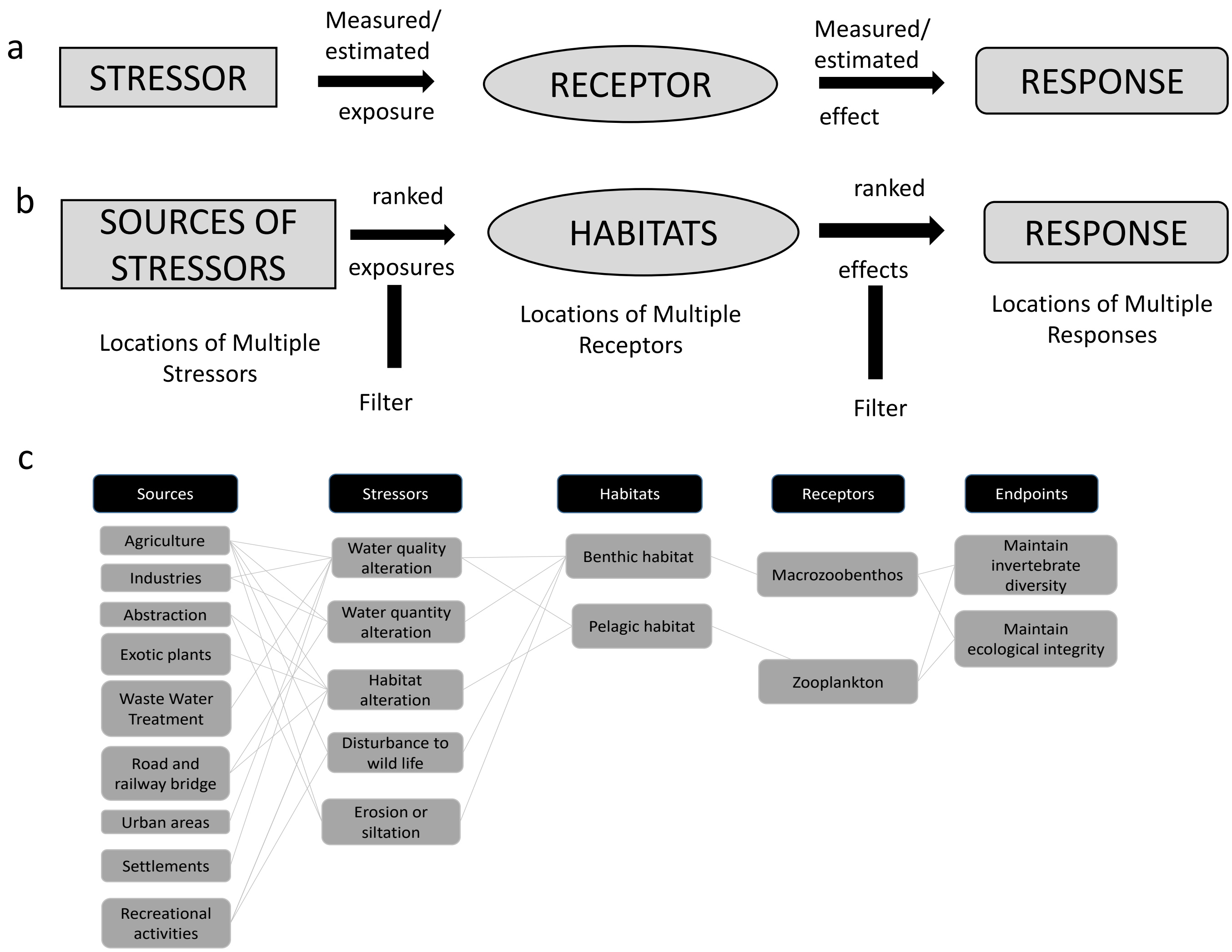

Fig. 4.2: a = traditional risk assessment, b = regional relative risk, c = conceptual model for invertebrates endpoint.

d

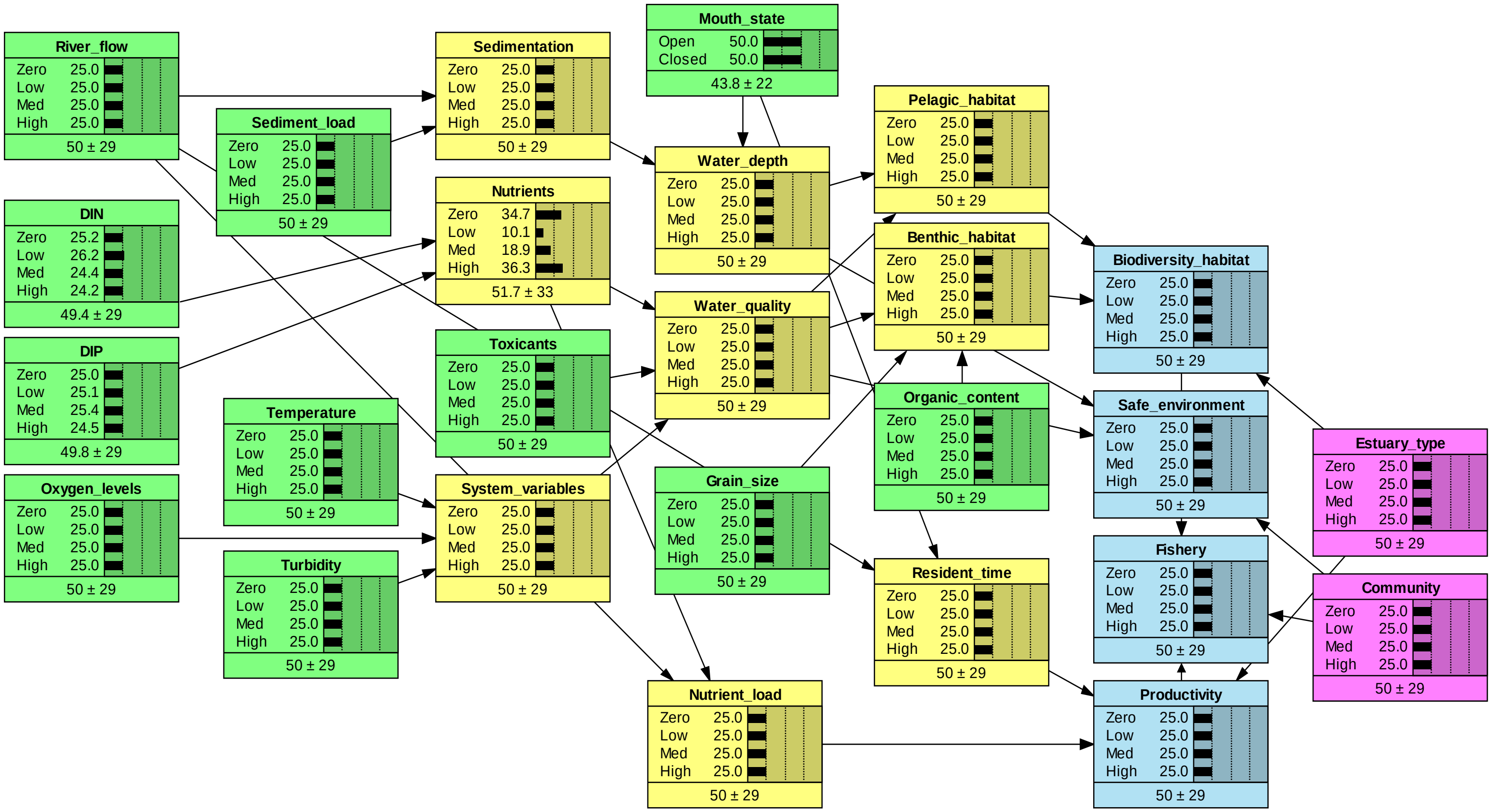

Fig. 4.3 (cont…): d = Bayesian network relative risk model- socio ecological endpoint

Conditional probability tables (CPTs) were completed for each intermediate and endpoint node to give description and quantification of relationships between two or more variables. Arrows or links in the Bayesian network were established based on known cause-effect pathways. During development of Bayesian networks, the connection of two or more parent nodes to a daughter node relied greatly on a CPT to quantify the relationship and for calculation of the distribution of the daughter node. Some variables may have more influence in the endpoint than the others and these can be identified through literature, in turn, the effect is reflected in the daughter node CPT with that parent node state having higher influence on the resulting state. The RRM is organized based on algebraic equations that sum up ranking for stressors, habitats and filters (Landis and Wiegers, 2005). Filters define the relationship between stressors, habitats and potential impact to the assessment endpoint (Fig. 4.2b). Filters are defined as numeric weighting criteria used to describe the relationship amongst the risk components and determine how the stressor and habitat are likely to overlap (i.e. exposure filter) and how likely they are to cause a certain effect (i.e. effect filter). A relative rank is assigned to both stressor and habitat and then filters are applied to examine the probability of exposure followed by probability of effect. The combination of ranks and weighting factors was done through multiplication and the product is the representation of a relative estimate of risk in a specific estuary. Final risk scores (RS) were then calculated for each estuary through multiplication of ranks with respective weighting factor (W_{ij}) as depicted below (Adapted from Landis and Wiegers, 2005).

RS= S_{ij} x H_{ik} x W_{jk}, where *i* = the risk region (estuary) series, *j* = the stressor series which include among others discharge (flows), nutrients and pollutants in the current study, *k* = the habitat series which is pelagic and benthic habitat in the current study, S_{ij} = rank chosen for the stressors between risk regions (estuaries), H_{ik} = rank chosen for the habitats between risk regions and W_{jk} = weighting factor established by the exposure or effect filter. Through summation of total RS for the stressors across all habitats, the relative risk resulting from a specific stressor can be calculated e.g. $RS_{stressor} = \sum (S_{ij} \times H_{ik} \times W_{jk})$ for *j* = 1 to *n*. Again, through summation of total RS for each habitat across all stressors, a potential total relative risk occurring within a given habitat can be calculated. E.g. $RS_{habitat} = \sum (S_{ij} \times H_{ik} \times W_{jk})$ for *k* = 1 to *n*. Netica software was used to calculate risk as it uses probabilistic inference to update the intermediate and endpoint nodes based on input probabilities and CPTs (Norsys Software, 2014). After the calculation of risk for the present scenario in the three estuaries studied, four more scenarios were proposed and risks were calculated for each scenario, endpoint, risk region (estuary) as well as flows (high/low) using Netica software. The additional scenarios were scenario 2 = 1995–2005, scenario 3 = pre major resource developments (<1950's), scenario 4 = 2025 with no mitigation or implementation of environmental protection legislation, scenario 5 = 2025 with mitigation and environmental protection legislation implemented.

4.2.6 Uncertainly analyses

Factors like scarcity of data in the study area, poor data quality, insufficient information on the relationships between components, lack of knowledge about how ecosystems function, omission of stressors and secondary effects may result in uncertainty in EcoRA (Obery and Landis, 2002; Yu et al., 2015). Uncertainty in risk predictions arise because risk predictions in RRM are point estimates derived from data and expert's subjective judgements. A series of uncertainty analyses were performed to provide primary insights into the model's behavior and performance and to provide insight on additional data gaps that might be present. Uncertainty analyses were performed to get an insight of potential effects that associated uncertainty in model assumptions, model parameters and model structure may have on the model findings. Monte Carlo uncertainty analysis estimates probability distribution for output variable by combining assigned probability distribution of input variables. We firstly assigned designations of zero, low, medium and high uncertainty to each endpoint and each scenario. By using Crystal ball® software as a macro in Microsoft Excel ® 2013, we ran the Monte Carlo simulations for 5000 iterations and output for each scenario, estuary and flows was derived. Uncertainty analyses performed in the current study were consistent with standard good practices in RRM analyses e.g. Landis (2005).

4.3 Results

4.3.1 Relative risk model results

Bayesian network-relative risk models for all endpoints, all scenarios, all estuaries and all flows are shown in supplementary material (S2). The RRM models show the probability distributions of input nodes, intermediate nodes and endpoint nodes. Relative risk scores and ranking information calculated during establishment of RRM models for all endpoints, scenarios, estuaries and flows was used to summarize results of the present study.

4.3.2 Overall risk to endpoints

To get an insight/trend of the overall risk scores to each endpoint in each estuary, each scenario and during different flows, the average risk scores and standard deviation calculated by Netica for each endpoint were used and this is presented in Fig. 4.3. Productivity generally displayed lower risk when compared with other endpoints in all estuaries, during all scenarios except for scenario 5, and during both low and high flows (Fig. 4.3). In scenario 5, productivity was most likely to be affected when compared with other endpoints in all estuaries and flows. Overall, scenario 3 which is a scenario before major resource development in the study area had the lowest scores of risk for all the endpoints with the Thukela Estuary displaying the lowest scores ranging from 20–24 (Fig. 4.3). The scenario that displayed highest risk scores to endpoints was scenario 4 which predicted risk scores to each endpoint for year 2025 if no laws and management measures are implemented (Fig. 4.3). The endpoint that is likely to be highly affected in scenario 4 is biodiversity habitat with scores ranging from 41 (uMvoti high flow) to 67 (uMvoti low flow) (Fig. 4.3). It is evident that if management measures and laws are implemented as depicted by scenario 5, the risk levels to the endpoints decrease, however the productivity endpoint during low flow will thus be likely to be affected, displaying risk scores ranging from 42 to 57 in Thukela and aMatikulu/Nyoni Estuary respectively (Fig. 4.3). In the present scenario (scenario 1) and scenario 2 the endpoint that is likely to be affected is fishery with risk scores ranging from 35 in Thukela Estuary during low flow to 55 in Thukela and uMvoti Estuary during high flow (Fig. 4.3). By summing all risk scores across all scenarios and flows we could compare risks to each endpoint for each entire estuary (Fig. 4.4). All endpoints selected for the current study were at highest risk in the uMvoti Estuary followed by the aMatikulu/Nyoni Estuary although the difference between the two estuaries was small with a difference of less than 20 across all endpoints (Fig. 4.4). The Thukela Estuary displayed lower risk for all endpoints with productivity endpoint being least affected (Fig. 4.4). Safe environment was most likely to be affected in the uMvoti with a highest score of 478, followed by fishery (475), biodiversity habitat (469) and productivity (463) (Fig. 4.4). In the Thukela Estuary biodiversity habitat was most likely to be affected with the highest score of 424 followed by safe environment (408), fishery (407) and productivity (356) (Fig. 4.4). Endpoints that are most likely to be affected in the aMatikulu/Nyoni Estuary are productivity and safe environment with both of them having a score of 461, followed by fishery (460) and biodiversity habitat (450) (Fig. 4.4).

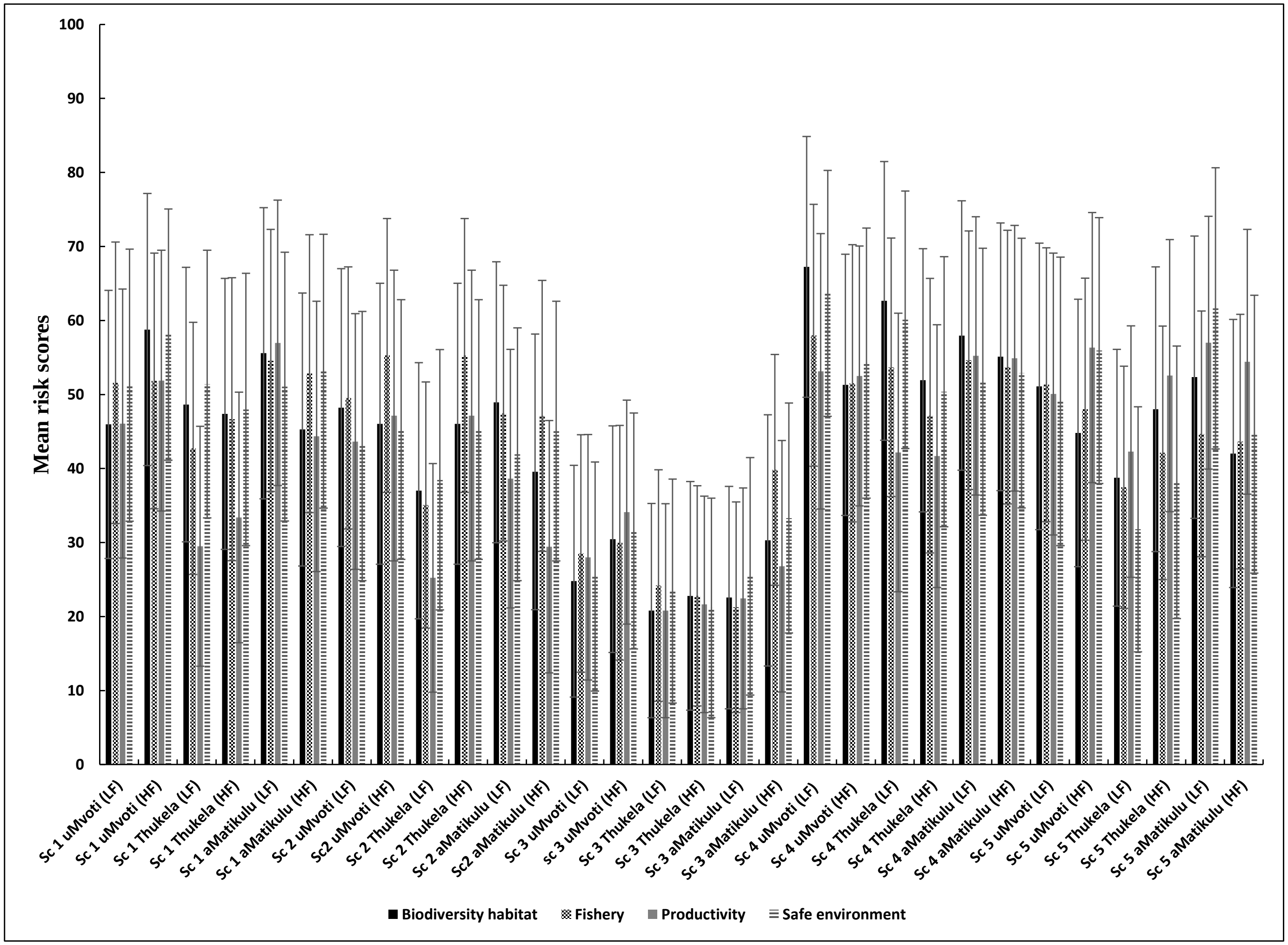

Fig. 4.4: Mean risk scores ±SD for endpoints in all estuaries, all scenarios (Sc) and different flows. Sc 1 = 2005–present, sc 2 = 1995–2005, sc 3 = pre major resource developments (<1950's), sc 4 = 2025 with no mitigation or implementation of environmental protection legislation, sc 5 = 2025 with mitigation and environmental protection legislation implemented, LF = low flow, HF = high flow.

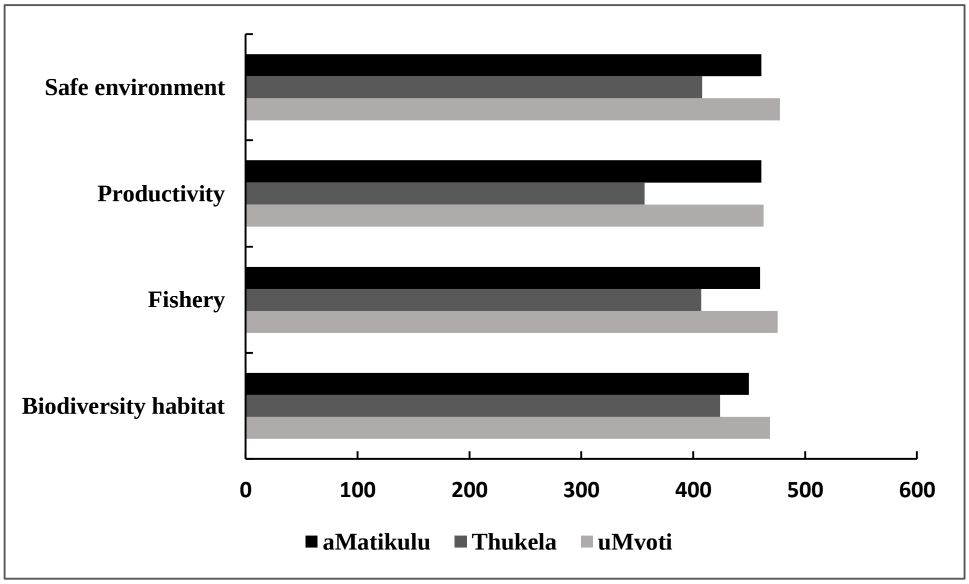

Fig. 4.5: Relative risk estimate for each endpoint in the uMvoti, Thukela and aMatikulu/Nyoni estuaries.

4.3.3 Risk state distribution of endpoints across all estuaries and scenarios

Risk state distribution for all endpoints across all estuaries, scenarios and flows are presented in Fig. 4.5. Endpoints generally displayed low to medium risk throughout all scenarios (except scenario 3) and flows (Fig 4.5). All endpoints generally displayed zero risk in scenario 3. Biodiversity habitat, productivity and safe environment are likely to be affected in scenario 4 as these endpoints show increase in risk distribution during this scenario. When we separated relative risk scores into high and low flows for each endpoint, some endpoints displayed higher risk scores during low flow when compared with high flow e.g. fishery and biodiversity habitat (Fig. 4.6). These endpoints generally displayed highest risk in scenario 4 followed by scenario 1 (Fig. 4.6). Productivity and safe environment did not show difference in risk distribution between high and low flow although the endpoints showed highest risk in scenario 4 followed by scenario 1 (Fig.4. 6).

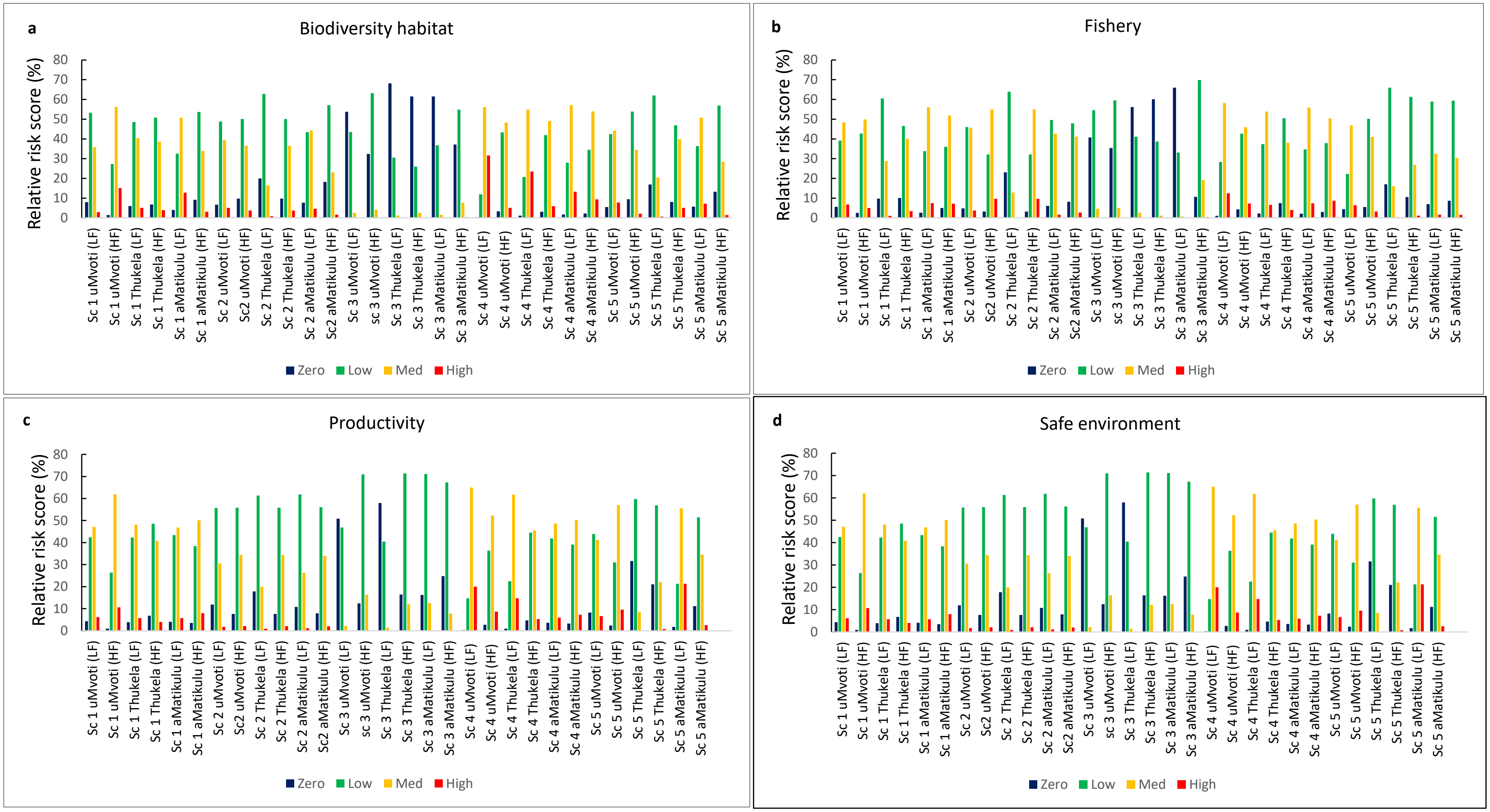

Fig. 4.6: Risk state distribution of endpoints across all estuaries, scenarios (sc) and flows. Sc 1 = 2005–present, sc 2 = 1995–2005, sc 3 = pre major resource developments (<1950's), sc 4 = 2025 with no mitigation or implementation of environmental protection legislation, sc 5 = 2025 with mitigation and environmental protection legislation implemented, LF = low flow, HF = high flow.

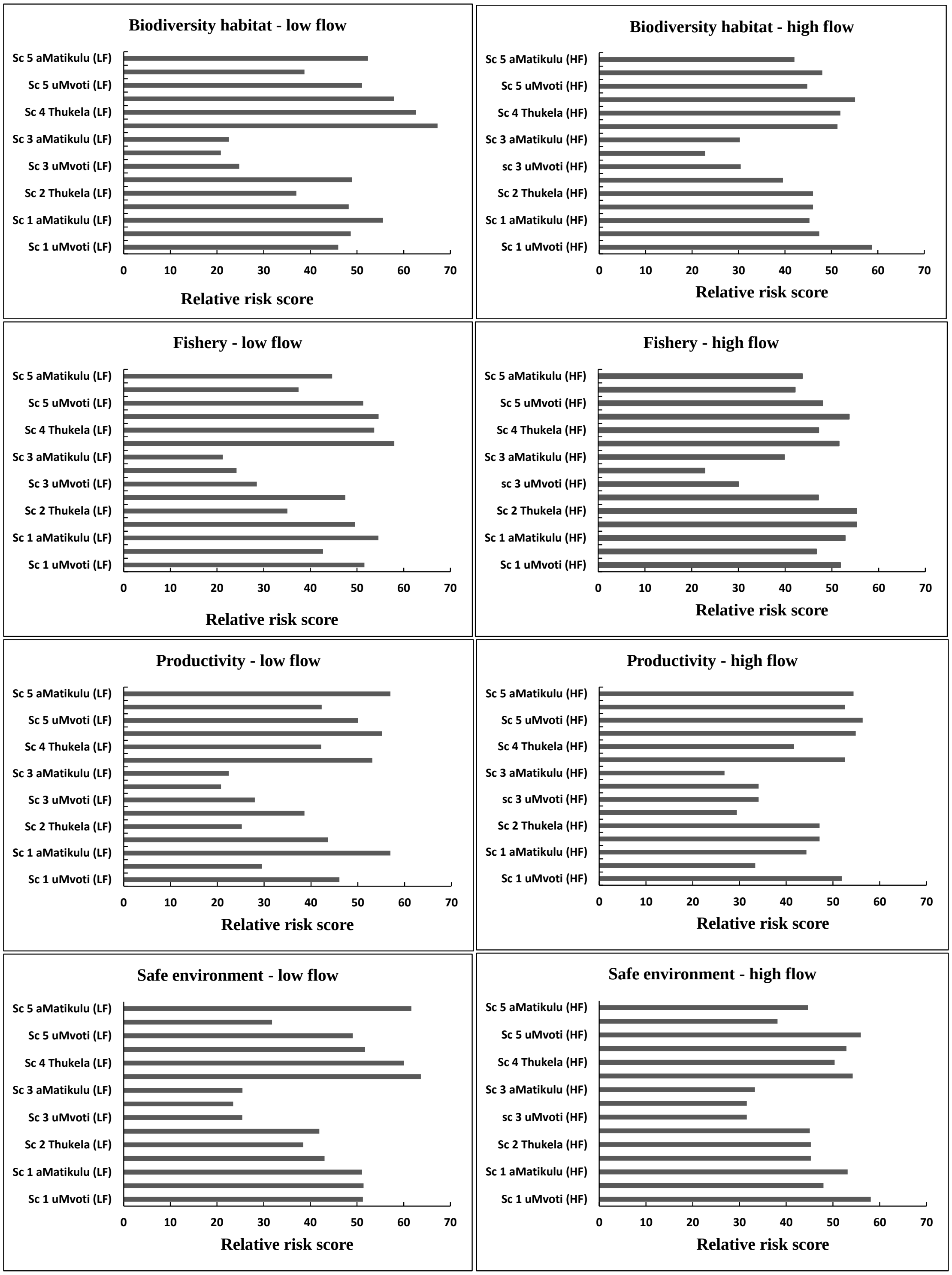

Fig. 4.7: Relative risk estimate of endpoints during low and high flow in all the scenarios (sc) and estuaries. Sc 1 = 2005–present, sc 2 = 1995–2005, sc 3 = pre major resource developments (<1950's), sc 4 = 2025 with no mitigation or implementation of environmental protection legislation, sc 5 = 2025 with mitigation and environmental protection legislation implemented, LF = low flow, HF = high flow.

4.3.4 Uncertainty analysis results

Major uncertainty in the current study lay in the spatial data with potential of heterogeneity of the land use, vegetation, and land cover of different slopes. The toxicity effects of chemicals entering estuaries derived from land use activities were not included in the current study e.g. pesticides from agriculture and polyaromatic hydrocarbons from urban land use. Other omitted data in the current study include toxicity measurements from multiple sources of waste water discharges. There were also uncertainties regarding the exposure pathways. The output distributions based on 5000 trials for both social and ecological endpoints in each estuary during both low and high flows are shown in supplementary material (S3). Generally, the uncertainty distributions of some social and ecological endpoints in the three estuaries had wider ranges which indicated high variability in risk estimates. Overall, the low flow periods showed vulnerability in the output of scenarios while high flow periods showed resilience with all scenarios showing relatively similar patterns. As expected, there was generally a high probability of endpoints to be at high risk during scenario 4, however after the implementation of mitigations and laws (i.e. Scenario 5) the probability for the endpoints to be at high risk was reduced. This was evident in the uMvoti and Thukela estuaries. Results of uncertainty analysis showed that the risk distributions of reference scenario which was scenario 3 fell within within zero to low risk while the rest of scenarios generally lay within low to medium risk scores. In the uMvoti and Thukela estuaries higher risk was associated with social endpoints indicating that people are at greater risk than the ecological components of these systems. However, in the aMatikulu/Nyoni Estuary, the ecological components of this system were at a greater risk as ecological endpoints displayed higher risk than the social endpoints. Scenario 1 showed relatively narrower ranges in all the estuaries indicating lower variability in risk estimates.

4.4 Discussion

Ecological risk assessment has been widely acknowledged as an important tool in improving environmental decision making (Yu et al., 2015). In the current study we conducted initial assessment of potential sources, stressors and effects in the uMvoti, Thukela and aMatikulu/Nyoni estuaries using the RRM. Results from the RRM provided useful information that can be used by the environmental managers in establishing management measures and controlling/minimizing dominant stressors and thus protecting or restoring important socio-ecological endpoints in these three estuaries. Overall risk results across all scenarios showed that all endpoints were at a highest risk in the uMvoti followed by aMatikulu/Nyoni Estuary while the Thukela displayed lower risk to all endpoints. Although the aMatikulu/Nyoni Estuary is in a relatively good ecological state, results of the current study showed that this system requires management focus just as uMvoti Estuary. The suggested reason for this is that if the endpoints in this system are at high risk, the sensitivity of this system will increase making it more vulnerable to external stressors. Risk state distribution results showed that endpoints across all estuaries and scenarios generally displayed low to medium risk with zero risk observed in scenario 3 in all endpoints. Risk state distribution results showed that if no management measures and no focus is being directed to mitigation, conservation or management the high risk state distribution will rise in most of the endpoints shifting the risk distribution from low/medium to high risk.

Results of the current study showed that some endpoints are at higher risk during low flow compared with high flow with fishery and biodiversity habitat being the examples. Such results further highlight that reduction in flows of these estuaries will further pose more stress to the endpoints. This further suggests that flow requirements for estuarine systems should be established and met so as to minimize risks levels posed by pre-identified stressors. Compared to uMvoti and Thukela estuaries, threats in the aMatikulu/Nyoni Estuary are minimal and are restricted to flow reduction and water quality alteration primarily. Sources of stressors in aMatikulu/Nyoni Estuary include one sugar mill and associated agricultural activities upstream of the aMatikulu River as well as effects of recent drought. In contrast, the uMvoti Estuary is largely affected by pollution from different anthropogenic land use sources further upstream including effluent input of treated sewage, sugar and paper mill effluents from Gledhow Sugar Mill and Sappi Stanger Mill, agricultural irrigation and several domestic uses by formal and informal settlements (places where people decide to live and build temporal shelters which usually have poor sanitation system. The Thukela system on the other hand is impacted by water abstraction for domestic use, industries, agriculture, mining, recreation, waste water treatment works, paper mill, and road and rail networks and these impacts dominate the lower reaches of the catchment. Such intense anthropogenic impacts are likely to pose risks to the endpoints of the Thukela and uMvoti estuarine systems and as previously reported such activities have caused some degradation in these systems. If no management measures are being implemented these systems will continue to deteriorate causing more stress in their ecological functioning. These impacts have also resulted in reduced oxygen levels in the uMvoti and Thukela estuaries during this study. Reduced oxygen levels can be one important stressor affecting the endpoints in the current study. Such trends further emphasize the need to look deeper at the sources of each stressor and their impacts into the entire ecosystem.

Relative risk model results indicated that endpoints selected for the present study are at risk and these results highlighted that if no management measures are being implemented, the risk level exposure will continue to increase to critical levels by 2025. Results of the presented study also showed that the uMvoti Estuary is currently highly impacted as this system displayed highest risk when compared with other estuaries. In support of the risk results of this study, water quality of the uMvoti system was described as grossly polluted since 1964 (Begg, 1978). The current study revealed that productivity as an endpoint was at lower risk when compared with other endpoints in all three estuaries except for scenario 5. The input nodes to this endpoint included nutrient load, resident time and estuary type. Low risk results to this endpoint suggests that nutrient levels and resident time in the three estuaries are not severely altered. This might also imply that enriched effluent comes out as less of an issue (risk). Establishment of different scenarios revealed that risk profiles generally change over time. Relative risk results showed that the endpoints in the three estuaries were at the lowest risk during scenario three representing a temporal period before major resource development. In addition, these results suggest that there has been a change in the risk of sources and stressors to the estuarine systems studied. Such change is likely to have been brought by the increase in human population, industrialization and agricultural plantations in the catchments of these estuaries. Scenario 4 (risk profile in year 2025 if no laws and management measures are implemented) displayed the highest risk to the endpoints. The endpoint likely to be at greatest risk in this scenario is biodiversity habitat in the uMvoti Estuary. It is evident that if management measures and laws are implemented as depicted by scenario 5, the risk to the endpoints decrease to the relatively acceptable levels, however the productivity endpoint during low flow will be thus likely to be affected, resulting in higher risk scores in Thukela and aMatikulu/Nyoni estuaries. These results suggest that measures should be taken to protect these systems and their endpoints. Efforts should be focused on protecting and restoring habitats to maintain the ecological integrity of theses estuaries, especially in the uMvoti

Estuary. In the present scenario and in scenario 2, the endpoint that is at highest risk is fishery with higher risk scores in the uMvoti and Thukela estuaries. These results continue to suggest that efforts should be focused in protecting and restoring estuarine habitats of species diversity with the uMvoti Estuary being the risk region of priority. From a management point of view, results of the present study suggest that uMvoti Estuary comprises substantial areas of valuable habitats that are at risk to multiple stressors.

The uMvoti Estuary has been rated severely degraded in terms of sedimentation and this condition deteriorates with time due to high sediment loads during flooding conditions (Badenhorst, 1990). At present, sedimentation issues have been raised as a concern in the aMatikulu/Nyoni Estuary although this system is in a relatively good state. Abnormally high volumes of sediments are present in the Thukela Estuary due to high sediment deposition coupled with reduced flows. These high sediment loads in this estuary have resulted in high turbidity levels in this system (current study). As historically reported, the Thukela Estuary continues to be threatened by sedimentation (Bosman et al., 2007). Disturbance of upstream vegetation associated with land use activities has a potential to increase sediment delivery, alter surface runoff and flows, increase lateral input of warm water, alter upstream infiltration irrespective of whether the catchment is urban, forested, agricultural or rangeland (Naiman et al., 1992). Such disturbances have been witnessed in the catchments of the uMvoti and Thukela Rivers (Venter, 2013). Apart from shifts in biological communities and change in morphology, alteration of riparian and upstream vegetation may also influence temperatures in the channel as well as stream assimilative capacity for heat (Poole and Berman, 2001). Water abstractions, channel engineering, removal of riparian vegetation and damming may also alter the ecological structure of these systems (Poole and Berman, 2001).

The current study confirmed that the RRM is an effective and rapid tool that can assist in environmental decision making in terms of ecological risk. The uncertainty results of the current study indicated that the RRM is sensitive to the uncertainties around the endpoints. Although these uncertainties depicted some limitations of the regional risk assessment framework and its application to the three river dominated estuaries of the current study, they did not affect the capacity of the RRM to give systematic means to evaluate and quantify the impacts of stressors to the endpoints on a relative basis which was the main goal of this study. In addition, the uncertainty results further supported the outcome of the overall risk distribution analysis to endpoints which depicted that in the uMvoti and Thukela estuaries, people are exposed to higher risk than the ecological components of these systems as the social endpoints were at higher risk than the ecological endpoints. Furthermore, the uncertainty results supported the relative risk distribution results when separated into flows as it highlighted that endpoints are more vulnerable during low flow when compared with high flow with fishery and biodiversity habitat being two examples. This suggests system vulnerability and sensitivity when flows are reduced, and further raise the need to look at the flow requirements of these systems and implementation of laws in order to maintain acceptable flows and maintain ecological integrity of these estuarine systems. The challenges in quantifying ecological impacts with complete certainty lies in difficulty of quantifying ecological factors. This is a result of complicated interrelations among various sources, stressors, habitats and endpoints at a regional scale. The main goal of EcoRA which is usually a semi-quantitative approach is to provide information for environmental management and to improve managers' decision making (Yu et al., 2015). One of the limitations in implementation of results/finding of RRM is the great uncertainty resulting from lack of data and poor quality of data (Yu et al., 2015). Additional data are thus required in the uMvoti, Thukela and aMatikulu/Nyoni estuaries in order to reduce uncertainty in the results. These data can be obtained through long term ecological monitoring of these systems so as to get the updates of the ecological states of these systems over time. It is suggested that the ecological important biotic and abiotic components of the systems be monitored, along with investigation of specific sources for different stressors. Such data will allow for better understanding of any impacts acting on the catchments of these systems. Extensive sampling will provide each risk region with specific data to inform model parameters. As data are becoming available, the relative risk models can be updated to depict new available knowledge of these estuarine systems. However, biomonitoring alone does not give an insight of what endpoints are at a high risk to be given management priorities. It is acknowledged that environmental monitoring and EcoRA are potentially complementary (Suter, 2001), where data essential to implement EcoRA are provided by environmental monitoring procedures. In addition, biomonitoring may also give insight about the accuracy of the previous EcoRAs and the effectiveness of risk-based environmental management (Suter, 2001). To further reduce uncertainty in future studies, CPTs can be customized to reflect the ecosystem specific dynamics of each estuary. This should be in an adaptive management context when additional data become available.

Future relative risk studies in the uMvoti, Thukela and aMatikulu/Nyoni systems should focus on individual stressors and their contributions to the overall risk in each estuary or risk region. Furthermore, evaluation of specific sources for specific stressors is also necessary. This can be better performed by categorizing stressors into physical, chemical and biological stressors. Such stressor breakdown will allow depiction and estimation of the contribution of stressors as percentage per risk region which will assist in environmental decision making. Although the preliminary RRM results were subject to uncertainty, the regional risk framework provided useful information to resource managers and planners regarding the types of stressors present together with their potential magnitude of risk they pose to the socio-ecological endpoints and to the entire estuarine ecosystems. Regional risk framework can thus be used as an adaptive management tool to assist in determination of uncertainties and data gaps and in provision of perspectives on data needs and research priorities for future management of a resource (Iannuzzi et al., 2009). Bayesian network RRM models established during the present study serve as a foundation for assessing the impacts for adaptive management strategies. The models can be used to calculate the conditions that result in the highest probability of meeting the management goals and predicting outcomes of management policies (Nyberg et al., 2006) Therefore, the upcoming regional risk framework studies in the three estuaries of the current study should incorporate the development of common data platforms for these estuaries, refinement of the RRM, identification of data gaps, intensive investigation into the sources of each stressor and prioritization of management and research alternatives. Remediation and mitigation focus should be directed to multiple stressors and this will have higher efficiency on reducing risk to endpoints of the uMvoti, Thukela and aMatikulu/Nyoni estuaries. In addition, it is necessary to consider all stressors to the endpoints not just those which initiated management action. Adherence to environmental laws for discharging waste water by industries and waste water treatment works should be strictly evaluated. Restoration of riparian vegetation of the estuaries studied can aid in improving water quality and aquatic habitats of these systems. Development of riparian buffers may be another important strategy to reduce sediment loading and erosion into these impacted estuaries. Estuary Management Plans are also urgently needed for these three estuaries so as to establish protection, conservation and management measures needed to minimise risk.

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4.7 Supplementary material (S1)

Table 4.1: Information on rank, scores and justification for each variable used in the current study.

Variable	Description of the variable	Measure (incl. units)	Rank	Score	Rank definition for variable	Rank definition for measure	Rank scheme uMvoti Estruary	Rank scheme Thukela Estruary	Rank scheme Amatikulu Estruary	Justification	References
River_flow (HIGH FLOW)	River inflow into the estuary meaured as discharge.	River discharge (m3/s)	Zero				>40	>68.19		In accordance with available information, the general effects of reduced volume of flows (discharge) into an estuary negatively affect the wellbeing of various ecosystem components and the services these systems provide to people (Allanson and Read 1995). During high flow more nutrients are introduced to the estuary from the river inflow. These elevated concentrations of nitrogen and phosphorus are from the surface runoff through leaching from weathering of rocks or from the agricultural plantations. However during extreme conditions of high flow (e.g. floods) these nutrients are diluted in the river and low concentrations are measured in an estuary. Extremely high flows (i.e. floods) may have a negative impact on community structures through the washing of phytoplankton and invertebrates to the adjacent sea. Reduced flows affect the river dominated state of the estuaries (Allanson and Read 1995). The river dominated/open mouth state of estuaries considered in this study is influenced by river flows which are impacted by land based activities and abstractions upstream. Where Reserve Determination studies for the estuaries considered in this case study are available, Reserve threshold discharge values for low flow have been used to represent the moderate-high threshold e.g. Thukela Estuary. In addition available threshold of potential concern values for low flow periods and Reserve values for high flow period have been used to represent the low-moderate threshold. The 50%tile discharge of natural flows was used to represent the zero-low threshold. Where no reserve flows were available the 50%tile, 75%tile and 100%tile was used to represent the zero-low, low-moderate and moderate-high thresholds respectively for the assessment.For determination of ranks in the uMvoti Estuary, the low flow (drought) values were selected to represent the threshold for moderate and high ranks (1.01 m3/s) (DWS, 2014). The 60th percentile was selected to represent moderate-low threshold (2.03 m3/s). For low-zero threshold, small freshet flows were selected to represent low-zero threshold (12 m3/s). For the high flow period, 60%ile was selected to represent the moderate-high threshold (2.03 m3/s). Small freshets were selected to represent moderate-low threshold (12 m3/s). Annual floods were selected to represent low-zero threshold (40 m3/s). There is no gauging station in the aMatikulu Estuary. Because the aMatikulu Estuary lies in the same eco-region as the uMvoti (-29. 26398, 31.03513), we used the flows of uMvoti Estuary as a proxy for aMatikulu Estuary. In this study discharge (in m3/s) was selected as the measure to represent the "Quantity" variable.	Allanson and Read 1995
			Low	50	Ideal operation area for use	Value (m3/s) representing threshold for 81-90%tite of natural base flows.	12-40	68.19-19.9			
			Moderate	75	Acceptable discharge but close to high	Value (m3/s) representing threshold for 90-99.9%tite of natural base flows.	2.03-12	19.8-9.2			
			High	100	Critical condition in flows	Flows below reserve requirements.	<2.03	<9.19			
River_flow (LOW FLOW)	River inflow into the estuary meaured as discharge.	River discharge (m3/s)	Zero			Flows close to natural/unaltered.	>12	>14.9		In accordance with available information, the general effects of reduced volume of flows (discharge) into an estuary is known to negatively affect the wellbeing of various ecosystem components and the services these systems provide to people (Allanson and Read 1995). Reduced flows as a result of drought or antropogenic activities (e.g. abstractions, damming and irrigation) lead to reduced concentrations of nutrients in estuaries as these system rely largely on nutrients from inland through surface runoff. In estuaries reduced flows are known to reduce nutrients levels and in turn primary productivity of these systems (Allanson and Read 1995). However, extremely high flows (i.e. floods) may have a negative impact on community structures through the washing of phytoplankton and invertebrates to the adjacent sea. Reduced flows affect the river dominated state of the estuaries (Allanson and Read 1995). The river dominated/open mouth state of the estuaries considered in this study is influenced by river flows which are impacted by land based activities and abstrations upstream. Where Reserve Determination studies for the estuaries considered in this case study are available, Reserve threshold discharge values for low flow have been used to represent the moderate-high threshold. In addition available threshold of potential concern values for low flow periods and Reserve values for high flow period have been used to represent the low-moderate threshold. The 50%tile discharge of natur0al flows was used to represent the zero-low threshold. Where no reserve flows were available the 50%tile, 75%tile and 100%tile was used to represent the zero-low, low-moderate and moderate-high thresholds respectively for the assessment. For determination of ranks in the uMvoti Estuary, the low flow (drought) values were selected to represent the threshold for moderate and high ranks (1.01 m3/s) (DWS, 2014). The 60th percentile was selected to represent moderate-low threshold (2.03 m3/s). For low-zero threshold, small freshet flows were selected to represent low-zero threshold (12 m3/s). For the high flow period, 60%ile was selected to represent the moderate-high threshold (2.03 m3/s). Small freshets were selected to represent moderate-low threshold (12 m3/s). Annual floods were selected to represent low-zero threshold (40 m3/s). There is no gauging station in the aMatikulu Estuary. Because the aMatikulu Estuary lies in the same eco-region as the uMvoti (-29. 26398, 31.03513), we used the flows of uMvoti Estuary as a proxy for aMatikulu Estuary. In this study discharge (in m3/s) was selected as the measure to represent the "Quantity" variable.	Allanson and Read 1995
			Low	50	Ideal operation area for use	Value (m3/s) representing threshold for 81-90%tite of natural base flows.	2.03-12	14.9-5.9			
			Moderate	75	Acceptable discharge but cloes to high	Value (m3/s) representing threshold for 90-99.9%tite of natural base flows.	1.01-2.03	5.9-4.8			
			High	100	Critical condition in flows	Flows below reserve requirements.	<1 .01	<4.8			
Mouth-state	Shape and size of the estuary	Size measured in area (km2)	Open	25	Permanently open estuary	Water volume needed is high.	<100 km^2	>100 km2	> 100 km2	The size and shape of an estuary determines many of its inherit physical features – tidal variation, retention time, responsiveness to flow and structural habitat features such as inter- and supratidal area. If the mouth of a big permanently open estuary closes high volume of water is required to open the mouth. If a mouth of a small estuary closes, less water volume is needed to breach and open the mouth.	DWA 2010
			Closed	50	temporarily open/closed estuary	Water volume needed is low.	<100 km^2	>100 km2	> 100 km2		

Table 4.1 (cont…): Information on rank, scores and justification for each variable used in the current study

Organic content	Percentage organic content in an estuary to maintain and support lowest components of the food web	%	Zero	25	Organic content high	Orgnic content can sufficiently support microbial communities on sedimets.	>4%	>4%	>4%	Particle size and organic content determine the cohesion of sediments, the understanding of which helps to predict response to flow changes. This detrital organic matter serves as a main energy source for microorganisms living in estuarine sediments. Sediment organic matter is derived from plant and animal detritus, bacteria or plankton formed in situ, or derived from natural and anthropogenic sources in catchments. Sewage and effluent from food-processing plants, pulp and paper mills and fish-farms are examples of organic-rich wastes of human origin. Bacteria quickly eat the less resistant molecules, such as the nucleic acids and many of the proteins. Sediment carbon and nutrient concentrations increase with decreasing grain size because organic matter adsorbs onto mineral surfaces and has a high affinity for fine-grained sediment. Decreased freshwater flows can alter the amount of organic matter that enters a coastal waterway and the rate at which it is flushed to the ocean	DWA 2010, Jorgensen and Boetius 2007, CSIRO 2000.
			Low	50	Organic content moderate	Organic content low but can still sustain microbial communities.	2-4%	2-4%	2-4%		
			Moderate	75	Organic content low	Moderate percentage of organic content with moderate to low support of microbial communitities.	0.5-2%	0.5-2%	0.5-2%		
			High	100	Organic content very low	Organic content very low that it can not sustain microbial communities.	<0.5%	<0.5%	<0.5%		
Sediment grain size	Sediment grain size is known to control the distribution of benthic invertebrates. Higher diversity is normally recorded in the fine to medium size sand and mud.	Sediment grain size measured in mm	Zero	25	Sediment grain size very fine	Sediment grain size fine and evenely sorted.	0.2 - 0.05 mm	0.2 - 0.05 mm	0.2 - 0.05 mm	Higher diversity is exhibited in the muddy habitat as well as fine and medium sized composition of sediment. These are considered as favourable habitat conditions. When the river is suffering from erosion processes the course sediment are likely to dominate the estuary bed. If there is very low flows the soft sediment accumulate in the estuary with no enough water to flush these sediments to the sea. As a result fine to medium sand as well as mud dominate the estuary. Well sorted sediment is known to suuport higher diversity, however poorly sorted sediment is known to support lower diversity.	Blair and McPherson 1999
			Low	50	sediment grain size medium	sediment grain size medium and evenly distributed.	0.5 - 0.2 mm	0.5 - 0.2 mm	0.5 - 0.2 mm		
			Moderate	75	Sediment grain size course	Sediment grain size course and evenly sorted.	2 - 0.5 mm	2 - 0.5 mm	2 - 0.5 mm		
			High	100	Sediment grain size very course	Sediment grain size very course and poorly sorted.	4 - 2 mm	4 - 2 mm	4 - 2 mm		
Temperature (High Flow)	Water temperature in the estuary influenced by both river inflow and tidal action	Water temperature measured in ^{o}C	Zero	25	No change in temperature	Pristine estuary. Catchment natural. No known activities which might alter temperature.				In South African estuaries temperature affects the growth and behavior of estuarine organisms, including predators like the blue crab, *Callinectes sapidus* . When the water gets too cold, the crab does not grow or molt at all, and goes into a physiological state called torpor, which is a kind of suspended animation. Temperature also affect the amount of oxygen that dissolves in water. This means that the amount of oxygen available to organisms is directly related to temperature. The more oxygen there is in the water, the healthier the ecosystem is. As temperature increases, the amount of oxygen dissolving in water decreases. High temperatures can be especially problematic for estuarine organisms because their oxygen demands generally increase with temperature. For each 10 °C rise in temperature, the rates of biological enzymatic processes often roughly double. Respiration rates of everything from bacteria to submerged aquatic vegetation to fish tend to follow this relationship as temperature increases. Unusually high temperatures, as might occur during an extended summer heat wave, can actually lead to death by “suffocation” of aquatic organisms.The temperature of the water also tells us what types of plants and animals are able to live in the estuary. All plants and animals have a range of temperatures in which they thrive. If the water in the estuary is outside the normal seasonal temperature range in which most estuarine organisms can comfortably live, it is probably an indication that something is adversely affecting the health of the estuary. Water temperature is known to affect the distribution of invertebrates in estuaries. Cooler waters are always observed in the lower reaches near the mouth area which is as a result of marine tidal influence. Species sensitive to temperature variations are recorded in low abundances in this region.	DWAF 2008, Taylor 1992, de Villiers and Hodgson 1999, Brylawski and Miller 2003.
			Low	50	Small change in temperature	Some minor anthropogenic changes to the river but no known changes to the natural temperature regime. Highly temperature sensitive species in lower abundance.	25-30	25-30	25-30		
			Moderate	75	Moderate change in temperature	Moderate change to temperature occurs infrequently. Most highly temperature sensitive species in lower abundance.	20-25 and 30-32	20-25 and 30-32	20-25 and 30-32		
			High	100	Serious change in temperature	Serious change to temperature regime occurs most of the time. Only species highly tolerant to temperature changes occur.	<20 or >32oC	<20 or >32oC	<20 or >32oC		

Table 4.1 (cont…): Information on rank, scores and justification for each variable used in the current study

Temperature (Low flow)	Water temperature in the estuary influenced by both river inflow and tidal action	Water temperature measured in ^{o}C	Zero	25	No change in temperature	Pristine estuary. Catchment natural. No known activities which might alter temperature.				In South African estuaries temperature affects the growth and behavior of estuarine organisms, including predators like the blue crab, Callinectes sapidus. When the water gets too cold, the crab does not grow or molt at all, and goes into a physiological state called torpor, which is a kind of suspended animation. Temperature also affect the amount of oxygen that dissolves in water. This means that the amount of oxygen available to organisms is directly related to temperature. The more oxygen there is in the water, the healthier the ecosystem is. As temperature increases, the amount of oxygen dissolving in water decreases. High temperatures can be especially problematic for estuarine organisms because their oxygen demands generally increase with temperature. For each 10 °C rise in temperature, the rates of biological enzymatic processes often roughly double. Respiration rates of everything from bacteria to submerged aquatic vegetation to fish tend to follow this relationship as temperature increases. Unusually high temperatures, as might occur during an extended summer heat wave, can actually lead to death by "suffocation" of aquatic organisms.The temperature of the water also tells us what types of plants and animals are able to live in the estuary. All plants and animals have a range of temperatures in which they thrive. If the water in the estuary is outside the normal seasonal temperature range in which most estuarine organisms can comfortably live, it is probably an indication that something is adversely affecting the health of the estuary. Water temperature is known to affect the distribution of invertebrates in estuaries. Cooler waters are always observed in the lower reaches near the mouth area which is as a result of marine tidal influence. Species sensitive to temperature variations are recorded in low abundances in this region.	DWAF 2008, Taylor 1992, de Villiers and Hodgson 1999, Brylawski and Miller 2003.
			Low	50	Small change in temperature	Some minor anthropogenic changes to the river but no known changes to the natural temperature regime. Highly temperature sensitive species in lower abundance.	25-30	25-30	25-30		
			Moderate	75	Moderate change in temperature	Moderate change to temperature occurs infrequently. Most highly temperature sensitive species in lower abundance.	20-25 and 30-32	20-25 and 30-32	20-25 and 30-32		
			High	100	Serious change in temperature	Serious change to temperature regime occurs most of the time. Only species highly tolerant to temperature changes occur.	<20 or >32oC	<20 or >32oC	<20 or >32oC		
Turbidity	The measure of water clarity or urbidity	Cloudiness of water measured in NTU	Zero	25	No change in water clarity	prisine estuary. No known man made modifications to the catchment. Changes in turbidity appears to be natural and related to natural catchment processes such as rainfall runoff.	1-4.9	1-4.9	1-4.9	Turbidity is the cloudiness or muddiness of water. In general the more suspended, solid material there is in water, the higher the water's turbidity and the lower its clarity. Many human activities in or near aquatic habitats resuspend bottom sediments and create turbid conditions that differ in scope, timing, duration, and intensity from the resuspension events induced by storms, freshets, or tidal flows. Suspended sediments can introduce a variety of responses from aquatic biota,mailnly because many attributes of the physical environment are affected (Wilber and Clarke 2001). Examples include, increased light attenuation caused by turbidity reduces visibility, shortens the depth of the photic zone, and can alter the vertical stratification of heat in the water column (Moore 1978). Not all these effects are detrimental to biota e.g. many fish thrive in turbid estuarine environments (Cyrus and Blaber, 1992; Blaber and Blaber 1980) presumably benefiting from a reduced risk of predation. The detrimental effects of increased concentrations of sediment include egg abrasion, reduced bivalve pumping rates, and direct mortality (Wilber and Clarke 2001). Suspended material can be particles of clay, silt, sand, algae, plankton and other substances. When turbidity is high there will be very low primary productivity in the estuary as a result of poor light penetration for photosynthesis. Such high turbidities may also cause reduced breeding and survival of fish and other aquatic animals (DWAF 2008).	DWAF 2008, Wilber and Clarke 2001, Cyrus and Blaber, 1992; Blaber and Blaber 1980, Moore 1978, Wilber and Clarke 2001
			Low	50	Small change in water clarity	Some minor man-made modifications to the catchment. Changes in turbidity appear to be largely natural.	5-9.9	5-9.9	5-9.9		
			Moderate	75	Moderate change clarity	Moderate changes to the catchment land-use resulted in unnaturally high sediment loads and high turbidity during runoff events. The impacts are however temporary.	10-20	10-20	10-20		
			High	100	Serious change in clarity	The catchment is known to have serious erosion problems, increased turbidity levels are present most of the time, large silt loads are deposited leading to a serious reduction in habitat. Low amounts of periphyton algae or phytoplankton are present.	>20	>20	>20		
Sediment load	Suspended sediments pollution affecting biota	Percentage land use cover measured (%)	Zero	25	Natural sediment load	Prisine estuary. No known man made modifications to the catchment. No known land use activities that might lead to erosion.	0-10	0-10	0-10	Availability of sediment and the ability of hydrodynamic forces to transport it provides the material for the construction of estuary barriers and the infilling of estuarine channels. There are two main sources of sediment including fluvial and marine sources (Cooper et al. 1999). Poor agricultural practises and deforestation also increase sediment load derived from land. Sediment availability determines wheather an estuary has a barrier or not. Most South African estuaries have a barrier that is composed of marine sediment (Cooper et al. 1999). Sediment stability of a system influences macrophytes colonisation. In systems where the sediment is constantly modified by dynamic processes, submerged macrophytes are absent (Adams et al. 1999). Suspended sediment loads in aquatic systems are known to affect aquatic ecological processes, e.g. reduced growth rate, decreased size, deoxygenation of water, trigger movement of fish, protection filter against predators (reduced visibility), reduced feeding efficiency (Wilber and Clarke 2001). Land degradation including loss of vegetation cover and increase in road and footpaths mays lead to higher surface runoff and therefore lower water infiltration and reduced baseflow. In addition, increased surface runoff on bare surfaces leads to increase in suspended sediment input in the river. The measure used in this case study is the percentage land use cover.	Wilber and Clarke 2001, Cooper et al. 1999, Adams et al. 1999, Wilber and Clarke 2001
			Low	50	Small change in sediment load	Some minor man-made modifications to the catchment. Changes in turbidity appear to be largely natural	10-15	10-15	10-15		
			Moderate	75	Moderate change in sediment load	Moderate changes to the catchment land-use resulted in unnaturally high sediment loads during runoff events. The impacts are however temporary.	15-50	15-50	15-50		
			High	100	Serious change in sediment load	The catchment is known to have serious erosion problems, increased sediment levels are present most of the time, large silt loads are deposited leading to a serious reduction in habitat. Low amounts of periphyton algae or phytoplankton are present.	50-100	50-100	50-100		

Table 4.1 (cont…): Information on rank, scores as well as justification for each variable used in the current study

DIP	Dissolved inorganic phosphorus	Dissolved inorganic phosphorus measured in mg/l	Zero	25	No change in natural concentrations	No known anthropogenic activities which migh change nutrient regime.	<0.005	<0.005	<0.005	Dissolved inorganic phosphorus is an essential element for all life forms. This nutrient plays a role in primary productivity in estuaries. Low concentrations of phosphorus may limit primary productivity while high concentrations (together with nitrogen) may lead to eutrophication. The origin of most nitrogen and phosphorus is through weathering of rocks and leaching of soils on land (Day 1981). At present, anthropogenic activities are the predominant source of nutrient elements in may aquatic ecosystems. Increased sewage discharge, agricultural fertilisation, and urbanisation have resulted in accelerated inflow of nutrients (Kennish 1986).	DWAF 2008, Day 1981, Kennish 1986
			Low	50	Small change in natural concentrations	Minor modification in the catchment affecting the nutrient status.	0.005-0.015	0.005-0.015	0.005-0.015		
			Moderate	75	Moderate chang in natural concentrations	Moderate modifications in the catchment affecting the nutrient status.	0.015-0.025	0.015-0.025	0.015-0.025		
			High	100	Serious change in natural concentrations	Serious modifications in the catchment affecting the nutrient status.	>0.125	>0.125	>0.125		
DIN	Dissolved inorganic nitrogen	Dissolved inorganic nitrogen measured in mg/l	Zero	25	No change in natural concentrations	No known anthropogenic activities which migh change nutrient regime.	<0.25	<0.25	<0.25	Dissolved inorganic nitrogen is an essential element for primary productivity in estuaries. Low concentrations of nitrogen may limity photosynthetic power of primary producers while high concentrations (together with phosphorus) may lead to eutrophication. The origin of most nitrogen and phosphorus is through weathering of rock and leaching of soils on land (Day 1981). At present, anthropogenic activities are the predominant source of nutrient elements in may aquatic ecosystems. Increased sewage discharge, agricultural fertilisation, and urbanisation have resulted in accelerated inflow of nutrients (Kennish 1986).	DWAF 2008, Day 1981, Kennish 1986
			Low	50	Small change in natural concentrations	Minor modification in the catchment affecting the nutrient status.	0.25-0.7	0.25-0.7	0.25-0.7		
			Moderate	75	Moderate chang in natural concentrations	Moderate modifications in the catchment affecting the nutrient status.	0.7-1	0.7-1	0.7-1		
			High	100	Serious change in natural concentrations	Serious modifications in the catchment affecting the nutrient status.	>4	>4	>4		
Oxygen	solved oxygen con	Oxygen measured in mg/l	Zero	25	No change from natural values	No known problem of concerns about dissolved oxygen. System prestine.	>8	>8	>8	To survive, aquatic animals must have sufficient levels of dissolved oxygen (DO) in the water. The amount of dissolved oxygen in an estuary's water is the major factor that determines the type and abundance of organisms that can live there. The mixing of surface waters by wind and waves increases the rate at which oxygen from the air can be dissolved or absorbed into the water. DO levels are influenced by temperature and salinity. The solubility of oxygen, or its ability to dissolve in water, decreases as the water's temperature and salinity increase. DO levels in an estuary also vary seasonally, with the lowest levels occurring during the late summer months when temperatures are highest. Bacteria, fungi, and other decomposer organisms reduce DO levels in estuaries because they consume oxygen while breaking down organic matter. Oxygen depletion may occur in estuaries when many plants die and decompose, or when wastewater with large amounts of organic material enters the estuary. In some estuaries, large nutrient inputs, typically from sewage, stimulate algal blooms. When the algae die, they begin to decompose. The process of decomposition depletes the surrounding water of oxygen and, in severe cases, leads to hypoxic (very low oxygen) conditions that kill aquatic animals. Shallow, well-mixed estuaries are less susceptible to this phenomenon because wave action and circulation patterns supply the waters with plentiful oxygen. Reduction in oxygen levels may result in altered ecological functioning of the estuary. This may include fish kills and invertebrate kills as a result of species which can't tolerate verly low oxygen levels.	DWAF 2008
			Low	50	Small change from natural values	Some man-made modifications in the catchment but no known problems or concerns about DO, most oxygen sensitive species are present.	7-8	7-8	7-8		
			Moderate	75	Moderate change from natural values	Some concerns about dissolved oxygen, some oxygen sensitive species are present but mostly oxygen tolerant species.	6-7	6-7	6-7		
			High	100	Serious change from natural values	Major know problems with low dissolved oxygen, anoxic odours sometimes present, only very low DO tolerant species present.	2-4	2-4	2-4		
Toxicity	Toxicants pollution from anthropogenic activity, i.e. toxic compounds, affecting the functioning of the system	Land covered with agriculture, industries and urban centres (% per risk region)	Zero	25	Toxicant input is very low, unmeasurable	Almost no land covered with agriculture, industries and urban centres.	0-10	0-10	0-10	Pollutants like toxic substances (e.g. chemicals and heavy metals), nutrients pollution (i.e. eutrophication) and pathogens (e.g. bacteria and viruses) have high impact on the health of an estuary. Untreated domestic wastewater discharge, small and large scale agriculture, untreated wastewater from industrial factories and wastewater from gold mines are other sources of toxicant input in estuaries. Toxicant input may in turn threaten human health if contaminated water and/or fish are consumed. If toxicants levels are high in water they can lead to death of aquatic organisms. If water or fish from polluted water is consumed in high amounts they can cause death to humans.	Jiang et al. 2006, Diop et al., 2015.
			Low	50	Toxicant input does not cause any adverse impact on estuarine fauna and flora as well as humans	Limited area covered with agriculture, industries and urban centress.	11-15	11-15	11-15		
			Moderate	75	Toxicant input causes tolerable adverse impact on estuarine fauna and flora as well as humans	Large area covered with agriculture, industries and urban centres.	16-50	16-50	16-50		
			High	100	Toxicant input causes unacceptable adverse impacton estuarine fauna and flora as well as humans	Large surface of land covered with agricultural and or rural/urban centers in the RR, resulting in increased toxicant levels in the water, threatening huamn health.	51-100	51-100	51-100		

Table 4.1 (cont…): Information on rank, scores as well as justification for each variable used in the current study

Estuary type	Mouth state and size of an estuary	surface area in km^2	Zero	25	Large estuary and high volume	Estuary large, river dominated, permanently open and drains high volume of water.	>3000 km^2	>3000 km^2	>1000 km2	The effect of stress or magnitude of an impact at a given time can be determined by the type and the size of an estuary. The equal amount of a pollutant e.g. mercury, in a given time will display different levels of impacts in different estuaries with larger estuaries being less sensitive when compared with smaller systems. River mouths generally drain large volumes of water and as a result their sensitivity is less than that of temporally/permanently open estuaries which drain lesser volumes.	
			Low	50	Large estuary and moderate volume	Estuary large, river dominated and drains moderate volume of water.	2000-3000 km^2	2000-3000 km^2	500-1000 km2		
			Moderate	75	Medium estuary and moderate volume	Estuary medium and drains moderate volume of water.	1000 km2	1000 km2	500		
			High	100	Small estuary and loe volume	Estuary small, temporarily open and drains low volume of water.	<100 km^2	<100 km^2	<100 km2		
Community	Human population size settled in the estuary catchment	Percentage of human population settling in estuarine catchments	Zero	25	Very small percentage of human population	Negligible percentage of human population settling in the estuarine catchment.	< 1 %	< 1 %	< 1 %	Human population settling near estuaries increase pressure in estuarine systems. Increasing population size together with industrialization are increasing sources of stresses in the estuarine systems which increase pollutants and nutrients in these systems.	
			Low	50	Small percentage of human population	Small percentage of human population settling in estuarine catchment.	10%	10%	10%		
			Moderate	75	Moderate percentage of human population	Moderate percentage of human population settling n the estuarinecatchment.	40%	40%	40%		
			High	100	High percentage of human population	High percentage of human population settling in estuarine catchment.	>50%	>50%	>50%		

Table 4.2: Conditional probability table for sedimentation as an intermediate node.

Sedimentation					
Sediment load	**River flow**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	91.21	8.57	0.11	0.11
Zero	Low	48.12	51.38	0.39	0.10
Zero	Moderate	7.73	82.22	9.95	0.10
Zero	High	0.40	50.09	49.12	0.40
Low	Zero	49.28	50.30	0.32	0.10
Low	Low	8.45	83.23	8.22	0.10
Low	Moderate	0.30	48.40	50.89	0.40
Low	High	0.10	8.46	82.96	8.47
Moderate	Zero	8.71	83.30	7.89	0.10
Moderate	Low	0.37	49.80	49.39	0.43
Moderate	Moderate	0.10	7.29	82.59	10.02
Moderate	High	0.10	0.33	51.20	48.37
High	Zero	0.11	21.62	75.45	2.83
High	Low	0.10	2.76	75.08	22.06
High	Moderate	0.10	0.20	35.31	64.39
High	High	0.11	0.11	8.74	91.03

Table 4.3: Conditional probability table for nutrients as an intermediate node.

Nutrients					
DIN	**DIP**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	90.26	9.52	0.11	0.11
Zero	Low	49.31	50.21	0.38	0.10
Zero	Moderate	9.04	82.61	8.25	0.10
Zero	High	0.12	20.64	76.23	3.01
Low	Zero	48.44	51.16	0.30	0.10
Low	Low	8.14	82.65	9.10	0.10
Low	Moderate	0.38	49.48	49.77	0.36
Low	High	0.10	2.78	76.31	20.82
Moderate	Zero	8.42	82.20	9.28	0.10
Moderate	Low	0.38	50.22	49.00	0.39
Moderate	Moderate	0.10	7.61	84.52	7.76
Moderate	High	0.10	0.19	40.16	59.55
High	Zero	0.40	48.20	51.00	0.39
High	Low	0.10	8.76	83.20	7.94
High	Moderate	0.10	0.32	48.92	50.66
High	High	0.11	0.11	10.02	89.75

Table 4.4: Conditional probability table for system variables as an intermediate node.

			System variables			
Temperature	Turbidity	Oxygen levels	Zero	Low	Moderate	High
Zero	Zero	Zero	92.52	7.27	0.11	0.11
Zero	Zero	Low	66.97	32.80	0.13	0.10
Zero	Zero	Moderate	31.23	67.85	0.83	0.10
Zero	Zero	High	0.59	61.81	37.43	0.17
Zero	Low	Zero	67.08	32.69	0.13	0.10
Zero	Low	Low	29.19	69.83	0.88	0.10
Zero	Low	Moderate	7.40	85.44	7.06	0.10
Zero	Low	High	0.12	31.23	67.73	0.91
Zero	Moderate	Zero	29.94	69.01	0.95	0.10
Zero	Moderate	Low	7.08	85.32	7.50	0.10
Zero	Moderate	Moderate	1.02	67.52	31.32	0.13
Zero	Moderate	High	0.10	10.08	84.48	5.34
Zero	High	Zero	2.05	79.10	18.74	0.11
Zero	High	Low	0.44	62.16	37.24	0.16
Zero	High	Moderate	0.16	38.20	61.09	0.54
Zero	High	High	0.11	18.66	78.90	2.33
Low	Zero	Zero	67.14	32.63	0.13	0.10
Low	Zero	Low	32.22	66.80	0.88	0.10
Low	Zero	Moderate	6.80	84.72	8.38	0.10
Low	Zero	High	0.13	32.57	66.44	0.85
Low	Low	Zero	33.06	66.12	0.72	0.10
Low	Low	Low	7.60	85.33	6.98	0.10
Low	Low	Moderate	0.88	69.07	29.92	0.13
Low	Low	High	0.10	11.38	83.86	4.66
Low	Moderate	Zero	7.52	84.76	7.61	0.10
Low	Moderate	Low	0.82	68.04	31.01	0.13
Low	Moderate	Moderate	0.12	30.33	68.88	0.67
Low	Moderate	High	0.10	2.28	77.92	19.70
Low	High	Zero	0.17	37.15	62.18	0.50
Low	High	Low	0.11	19.19	78.61	2.09
Low	High	Moderate	0.10	7.38	85.65	6.87
Low	High	High	0.10	2.46	77.92	19.52
Moderate	Zero	Zero	30.69	68.32	0.90	0.10
Moderate	Zero	Low	7.70	84.71	7.49	0.10
Moderate	Zero	Moderate	0.72	66.41	32.74	0.12
Moderate	Zero	High	0.10	10.51	84.45	4.94
Moderate	Low	Zero	7.74	84.67	7.49	0.10
Moderate	Low	Low	0.94	68.73	30.20	0.13
Moderate	Low	Moderate	0.13	31.50	67.31	1.06
Moderate	Low	High	0.10	2.00	78.99	18.91
Moderate	Moderate	Zero	0.91	68.83	30.11	0.15
Moderate	Moderate	Low	0.13	30.59	68.14	1.14
Moderate	Moderate	Moderate	0.10	7.16	85.21	7.53
Moderate	Moderate	High	0.10	0.39	51.79	47.72
Moderate	High	Zero	0.10	7.64	84.71	7.55
Moderate	High	Low	0.10	2.28	77.39	20.22
Moderate	High	Moderate	0.10	0.52	59.61	39.78
Moderate	High	High	0.10	0.16	40.47	59.27
High	Zero	Zero	0.10	0.25	38.53	61.13
High	Zero	Low	0.10	0.21	35.43	64.26
High	Zero	Moderate	0.10	0.15	29.72	70.03
High	Zero	High	0.10	0.12	23.13	76.64
High	Low	Zero	0.10	0.18	33.30	66.42
High	Low	Low	0.11	0.17	29.48	70.25
High	Low	Moderate	0.10	0.12	21.82	77.96
High	Low	High	0.11	0.12	19.75	80.02
High	Moderate	Zero	0.10	0.14	29.72	70.04
High	Moderate	Low	0.10	0.12	22.36	77.42
High	Moderate	Moderate	0.11	0.12	17.40	82.38
High	Moderate	High	0.11	0.11	14.35	85.44
High	High	Zero	0.10	0.12	22.31	77.48
High	High	Low	0.11	0.12	17.51	82.27
High	High	Moderate	0.11	0.11	14.85	84.93
High	High	High	0.11	0.11	12.40	87.38

Table 4.5: Conditional probability table for water depth as an intermediate node.

Water depth					
Mouth state	**Sedimentation**	**Zero**	**Low**	**Moderate**	**High**
Open	Zero	68.90	30.77	0.23	0.10
Open	Low	27.99	67.37	4.53	0.10
Open	Moderate	5.03	68.01	26.73	0.23
Open	High	0.25	28.61	66.74	4.40
Closed	Zero	8.02	82.36	9.53	0.10
Closed	Low	0.38	51.39	47.86	0.36
Closed	Moderate	0.10	8.11	83.63	8.16
Closed	High	0.10	0.38	48.72	50.80

Table 4.6: Conditional probability table for water quality as an intermediate node.

			Water quality			
Nutrients	**Toxicants**	**System variables**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	Zero	91.54	8.24	0.11	0.11
Zero	Zero	Low	68.96	30.81	0.13	0.10
Zero	Zero	Moderate	30.97	67.99	0.94	0.10
Zero	Zero	High	18.57	79.37	1.96	0.10
Zero	Low	Zero	68.63	31.13	0.13	0.10
Zero	Low	Low	31.55	67.66	0.68	0.10
Zero	Low	Moderate	7.11	85.44	7.35	0.10
Zero	Low	High	4.79	84.27	10.83	0.10
Zero	Moderate	Zero	30.81	68.13	0.96	0.10
Zero	Moderate	Low	7.38	85.14	7.38	0.10
Zero	Moderate	Moderate	0.78	67.73	31.36	0.13
Zero	Moderate	High	0.80	67.69	31.38	0.13
Zero	High	Zero	0.10	0.29	40.14	59.47
Zero	High	Low	0.10	0.19	33.60	66.11
Zero	High	Moderate	0.10	0.15	30.61	69.14
Zero	High	High	0.10	0.12	22.90	76.87
Low	Zero	Zero	69.40	30.37	0.13	0.10
Low	Zero	Low	30.25	68.67	0.97	0.11
Low	Zero	Moderate	6.77	85.77	7.36	0.10
Low	Zero	High	1.27	73.34	25.27	0.11
Low	Low	Zero	30.21	68.84	0.86	0.10
Low	Low	Low	6.74	85.42	7.75	0.10
Low	Low	Moderate	0.88	67.06	31.92	0.14
Low	Low	High	0.25	45.77	53.59	0.39
Low	Moderate	Zero	7.11	85.32	7.47	0.10
Low	Moderate	Low	0.90	66.60	32.37	0.13
Low	Moderate	Moderate	0.13	30.92	67.99	0.97
Low	Moderate	High	0.11	20.07	77.88	1.95
Low	High	Zero	0.10	0.17	33.37	66.36
Low	High	Low	0.10	0.14	27.62	72.14
Low	High	Moderate	0.11	0.12	22.18	77.59
Low	High	High	0.11	0.12	18.73	81.04
Moderate	Zero	Zero	31.26	67.76	0.88	0.10
Moderate	Zero	Low	6.86	84.19	8.85	0.10
Moderate	Zero	Moderate	0.94	69.09	29.84	0.13
Moderate	Zero	High	0.12	24.54	74.13	1.21
Moderate	Low	Zero	8.42	83.43	8.05	0.10
Moderate	Low	Low	0.78	68.15	30.93	0.14
Moderate	Low	Moderate	0.12	32.38	66.65	0.85
Moderate	Low	High	0.10	7.41	84.65	7.84
Moderate	Moderate	Zero	0.89	67.79	31.18	0.13
Moderate	Moderate	Low	0.13	31.22	67.72	0.93
Moderate	Moderate	Moderate	0.10	6.78	85.50	7.62
Moderate	Moderate	High	0.10	1.27	73.11	25.52
Moderate	High	Zero	0.10	0.15	26.27	73.48
Moderate	High	Low	0.10	0.13	23.80	75.97
Moderate	High	Moderate	0.10	0.12	17.88	81.90
Moderate	High	High	0.11	0.11	14.69	85.09
High	Zero	Zero	39.38	59.98	0.54	0.10
High	Zero	Low	8.54	83.46	7.90	0.10
High	Zero	Moderate	0.63	60.95	38.23	0.19
High	Zero	High	0.11	18.69	79.05	2.15
High	Low	Zero	7.31	85.32	7.27	0.10
High	Low	Low	0.57	60.54	38.69	0.19
High	Low	Moderate	0.11	20.59	77.25	2.05
High	Low	High	0.10	2.06	76.91	20.93
High	Moderate	Zero	0.54	61.00	38.29	0.16
High	Moderate	Low	0.11	19.66	78.39	1.84
High	Moderate	Moderate	0.10	2.05	79.70	18.14
High	Moderate	High	0.10	0.14	39.32	60.43
High	High	Zero	0.10	0.12	24.32	75.45
High	High	Low	0.10	0.12	19.83	79.95
High	High	Moderate	0.11	0.11	14.08	85.71
High	High	High	0.11	0.12	11.62	88.15

Table 4.7: Conditional probability table for nutrient load as an intermediate node.

		Nutrient load			
Nutrients	**River flow**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	90.38	9.40	0.11	0.11
Zero	Low	49.89	49.66	0.35	0.10
Zero	Moderate	8.56	82.79	8.55	0.10
Zero	High	0.37	51.19	48.04	0.39
Low	Zero	49.87	49.67	0.36	0.10
Low	Low	7.57	83.55	8.77	0.10
Low	Moderate	0.33	47.70	51.69	0.27
Low	High	0.10	9.32	81.96	8.62
Moderate	Zero	7.65	84.29	7.96	0.10
Moderate	Low	0.33	50.03	49.18	0.46
Moderate	Moderate	0.10	8.09	83.35	8.46
Moderate	High	0.10	0.40	50.20	49.31
High	Zero	0.38	50.77	48.50	0.35
High	Low	0.10	9.04	82.51	8.34
High	Moderate	0.10	0.32	49.89	49.69
High	High	0.11	0.11	8.87	90.91

Table 4.8: Conditional probability table for pelagic habitat as an intermediate node.

		Pelagic habitat			
Water depth	**Water quality**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	90.79	8.99	0.11	0.11
Zero	Low	49.95	49.58	0.38	0.10
Zero	Moderate	8.65	82.77	8.49	0.10
Zero	High	0.40	50.08	49.22	0.30
Low	Zero	49.22	50.36	0.32	0.10
Low	Low	6.85	84.04	9.01	0.10
Low	Moderate	0.31	49.55	49.81	0.33
Low	High	0.10	9.41	83.05	7.44
Moderate	Zero	9.01	82.79	8.10	0.10
Moderate	Low	0.34	50.64	48.61	0.41
Moderate	Moderate	0.10	8.26	83.38	8.26
Moderate	High	0.10	0.39	50.91	48.60
High	Zero	0.10	0.32	44.93	54.65
High	Low	0.10	0.16	29.29	70.45
High	Moderate	0.11	0.12	20.26	79.51
High	High	0.11	0.11	11.42	88.36

Table 4.9: Conditional probability table for benthic habitat as an intermediate node.

			Benthic habitat			
Water quality	**Grain size**	**Organic content**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	Zero	91.61	8.17	0.11	0.11
Zero	Zero	Low	68.54	31.23	0.13	0.10
Zero	Zero	Moderate	32.21	66.99	0.70	0.10
Zero	Zero	High	39.21	60.01	0.68	0.10
Zero	Low	Zero	70.25	29.50	0.14	0.10
Zero	Low	Low	30.10	68.89	0.90	0.10
Zero	Low	Moderate	8.26	84.46	7.17	0.10
Zero	Low	High	2.82	78.42	18.65	0.11
Zero	Moderate	Zero	30.55	68.47	0.88	0.10
Zero	Moderate	Low	7.59	85.37	6.94	0.10
Zero	Moderate	Moderate	0.73	67.53	31.61	0.13
Zero	Moderate	High	0.11	20.56	76.79	2.53
Zero	High	Zero	29.03	69.71	1.16	0.10
Zero	High	Low	11.36	83.25	5.29	0.10
Zero	High	Moderate	3.04	79.86	16.99	0.11
Zero	High	High	0.10	0.71	58.54	40.66
Low	Zero	Zero	68.44	31.33	0.13	0.10
Low	Zero	Low	31.01	67.88	1.01	0.10
Low	Zero	Moderate	7.37	85.69	6.84	0.10
Low	Zero	High	18.87	78.95	2.09	0.10
Low	Low	Zero	32.69	66.29	0.92	0.10
Low	Low	Low	7.48	85.24	7.18	0.10
Low	Low	Moderate	0.88	67.76	31.23	0.12
Low	Low	High	0.57	59.95	39.30	0.18
Low	Moderate	Zero	7.69	84.38	7.83	0.10
Low	Moderate	Low	0.75	67.94	31.17	0.14
Low	Moderate	Moderate	0.14	31.61	67.41	0.83
Low	Moderate	High	0.10	8.33	82.94	8.63
Low	High	Zero	1.93	76.22	21.74	0.11
Low	High	Low	0.41	53.66	45.63	0.30
Low	High	Moderate	0.13	28.77	70.10	1.01
Low	High	High	0.10	0.18	40.86	58.85
Moderate	Zero	Zero	31.04	68.07	0.79	0.10
Moderate	Zero	Low	6.93	85.91	7.05	0.10
Moderate	Zero	Moderate	0.78	68.65	30.44	0.12
Moderate	Zero	High	8.11	83.28	8.51	0.10
Moderate	Low	Zero	6.64	85.42	7.84	0.10
Moderate	Low	Low	0.84	67.35	31.69	0.12
Moderate	Low	Moderate	0.13	31.76	67.07	1.04
Moderate	Low	High	0.17	39.19	60.09	0.55
Moderate	Moderate	Zero	1.09	66.89	31.90	0.12
Moderate	Moderate	Low	0.12	30.84	68.23	0.81
Moderate	Moderate	Moderate	0.10	7.83	84.82	7.26
Moderate	Moderate	High	0.10	2.19	76.75	20.96
Moderate	High	Zero	0.11	23.64	74.61	1.65
Moderate	High	Low	0.10	7.57	83.75	8.58
Moderate	High	Moderate	0.10	1.68	74.55	23.67
Moderate	High	High	0.10	0.11	20.01	79.78
High	Zero	Zero	58.07	41.63	0.20	0.10
High	Zero	Low	5.21	79.22	15.47	0.10
High	Zero	Moderate	0.11	15.56	79.80	4.53
High	Zero	High	0.10	0.19	38.20	61.52
High	Low	Zero	45.62	53.92	0.36	0.10
High	Low	Low	1.68	71.77	26.41	0.14
High	Low	Moderate	0.10	8.81	82.20	8.89
High	Low	High	0.11	0.13	27.88	71.89
High	Moderate	Zero	31.53	67.09	1.28	0.10
High	Moderate	Low	0.71	60.06	39.03	0.21
High	Moderate	Moderate	0.10	4.27	80.93	14.71
High	Moderate	High	0.10	0.11	16.23	83.56
High	High	Zero	20.36	76.71	2.83	0.10
High	High	Low	0.34	47.63	51.61	0.42
High	High	Moderate	0.10	2.22	72.67	25.00
High	High	High	0.11	0.11	10.40	89.38

Table 4.10: Conditional probability table for resident time as an intermediate node.

		Resident time			
Mouth state	**River flow**	**Zero**	**Low**	**Moderate**	**High**
Open	Zero	68.83	30.77	0.30	0.10
Open	Low	28.17	67.65	4.08	0.10
Open	Moderate	3.90	67.09	28.80	0.21
Open	High	0.60	39.32	57.36	2.72
Closed	Zero	7.47	82.92	9.52	0.10
Closed	Low	0.37	48.13	51.04	0.45
Closed	Moderate	0.10	7.82	83.11	8.97
Closed	High	0.10	0.52	56.87	42.50

Table 4.11: Conditional probability table for biodiversity habitat as an endpoint node.

			Biodiversity habitat			
Pelagic habitat	Benthic habitat	Estuary type	Zero	Low	Moderate	High
Zero	Zero	Zero	92.67	7.11	0.11	0.11
Zero	Zero	Low	67.15	32.62	0.13	0.10
Zero	Zero	Moderate	31.45	67.54	0.91	0.10
Zero	Zero	High	7.49	84.71	7.70	0.10
Zero	Low	Zero	67.88	31.89	0.13	0.10
Zero	Low	Low	31.17	68.00	0.73	0.10
Zero	Low	Moderate	6.84	85.71	7.35	0.10
Zero	Low	High	0.87	67.51	31.49	0.12
Zero	Moderate	Zero	32.14	67.00	0.76	0.10
Zero	Moderate	Low	6.77	85.80	7.33	0.10
Zero	Moderate	Moderate	0.83	67.65	31.38	0.14
Zero	Moderate	High	0.13	31.04	68.07	0.76
Zero	High	Zero	7.10	85.28	7.52	0.10
Zero	High	Low	0.74	65.64	33.49	0.12
Zero	High	Moderate	0.13	33.98	65.04	0.85
Zero	High	High	0.10	7.01	86.05	6.84
Low	Zero	Zero	67.57	32.19	0.14	0.10
Low	Zero	Low	30.00	69.02	0.88	0.10
Low	Zero	Moderate	8.54	83.99	7.37	0.10
Low	Zero	High	0.84	69.74	29.29	0.13
Low	Low	Zero	30.45	68.69	0.76	0.10
Low	Low	Low	6.68	86.10	7.12	0.10
Low	Low	Moderate	0.69	66.35	32.83	0.13
Low	Low	High	0.14	31.90	67.11	0.86
Low	Moderate	Zero	7.91	83.87	8.12	0.10
Low	Moderate	Low	0.72	67.43	31.72	0.13
Low	Moderate	Moderate	0.14	30.99	68.03	0.84
Low	Moderate	High	0.10	7.60	85.37	6.94
Low	High	Zero	0.77	67.77	31.34	0.12
Low	High	Low	0.14	31.19	67.78	0.89
Low	High	Moderate	0.10	7.17	85.23	7.50
Low	High	High	0.10	0.80	68.60	30.50
Moderate	Zero	Zero	30.76	68.39	0.75	0.10
Moderate	Zero	Low	8.00	85.17	6.73	0.10
Moderate	Zero	Moderate	0.85	67.83	31.17	0.15
Moderate	Zero	High	0.15	29.29	69.61	0.95
Moderate	Low	Zero	7.02	85.83	7.04	0.10
Moderate	Low	Low	0.65	68.38	30.84	0.14
Moderate	Low	Moderate	0.12	32.50	66.71	0.67
Moderate	Low	High	0.10	7.69	85.50	6.72
Moderate	Moderate	Zero	0.86	67.83	31.18	0.14
Moderate	Moderate	Low	0.13	32.16	67.08	0.64
Moderate	Moderate	Moderate	0.10	7.54	85.09	7.28
Moderate	Moderate	High	0.10	1.04	68.02	30.84
Moderate	High	Zero	0.14	31.22	67.85	0.80
Moderate	High	Low	0.10	7.53	85.72	6.64
Moderate	High	Moderate	0.10	0.74	65.39	33.78
Moderate	High	High	0.10	0.12	32.29	67.49
High	Zero	Zero	7.28	85.12	7.50	0.10
High	Zero	Low	0.78	66.87	32.22	0.13
High	Zero	Moderate	0.13	31.96	67.08	0.83
High	Zero	High	0.10	7.79	84.51	7.60
High	Low	Zero	0.78	68.09	31.01	0.12
High	Low	Low	0.14	30.10	68.90	0.85
High	Low	Moderate	0.10	7.40	85.09	7.40
High	Low	High	0.10	0.86	67.66	31.39
High	Moderate	Zero	0.12	30.52	68.50	0.86
High	Moderate	Low	0.10	7.91	84.55	7.44
High	Moderate	Moderate	0.10	0.90	68.58	30.42
High	Moderate	High	0.10	0.14	32.51	67.25
High	High	Zero	0.10	6.72	84.88	8.29
High	High	Low	0.10	0.95	68.88	30.07
High	High	Moderate	0.10	0.13	33.02	66.74
High	High	High	0.11	0.11	8.54	91.24

Table 4.12: Conditional probability table for safe environment as an endpoint node.

Safe environment						
Water depth	**Water quality**	**Community**	**Zero**	**Low**	**Moderate**	**High**
Zero	Zero	Zero	87.65	12.13	0.11	0.11
Zero	Zero	Low	60.62	39.07	0.20	0.10
Zero	Zero	Moderate	16.26	81.03	2.62	0.10
Zero	Zero	High	1.72	74.81	23.37	0.11
Zero	Low	Zero	57.66	42.04	0.20	0.10
Zero	Low	Low	17.18	79.79	2.93	0.10
Zero	Low	Moderate	1.42	73.32	25.15	0.11
Zero	Low	High	0.14	33.30	65.70	0.85
Zero	Moderate	Zero	17.23	79.33	3.34	0.10
Zero	Moderate	Low	1.74	72.95	25.20	0.12
Zero	Moderate	Moderate	0.14	30.71	68.22	0.94
Zero	Moderate	High	0.10	4.95	84.06	10.88
Zero	High	Zero	1.94	73.33	24.62	0.11
Zero	High	Low	0.14	32.51	66.59	0.76
Zero	High	Moderate	0.10	4.70	82.82	12.38
Zero	High	High	0.10	0.33	49.34	50.23
Low	Zero	Zero	81.86	17.92	0.11	0.11
Low	Zero	Low	39.45	59.86	0.58	0.10
Low	Zero	Moderate	7.68	84.24	7.98	0.10
Low	Zero	High	0.41	58.34	41.05	0.19
Low	Low	Zero	41.96	57.44	0.50	0.10
Low	Low	Low	7.11	84.87	7.92	0.10
Low	Low	Moderate	0.50	58.95	40.40	0.15
Low	Low	High	0.10	16.24	80.67	2.99
Low	Moderate	Zero	7.86	84.27	7.77	0.10
Low	Moderate	Low	0.63	60.83	38.37	0.18
Low	Moderate	Moderate	0.10	16.47	80.61	2.82
Low	Moderate	High	0.10	1.76	75.20	22.95
Low	High	Zero	0.47	57.34	41.99	0.20
Low	High	Low	0.10	17.12	79.88	2.89
Low	High	Moderate	0.10	1.51	75.05	23.34
Low	High	High	0.10	0.13	32.22	67.55
Moderate	Zero	Zero	68.28	31.49	0.13	0.10
Moderate	Zero	Low	23.28	75.03	1.60	0.10
Moderate	Zero	Moderate	3.30	78.87	17.73	0.10
Moderate	Zero	High	0.19	39.43	59.96	0.43
Moderate	Low	Zero	24.33	73.95	1.62	0.10
Moderate	Low	Low	2.59	79.71	17.60	0.10
Moderate	Low	Moderate	0.20	40.65	58.61	0.55
Moderate	Low	High	0.10	8.01	83.52	8.37
Moderate	Moderate	Zero	3.07	80.97	15.85	0.11
Moderate	Moderate	Low	0.20	42.02	57.36	0.42
Moderate	Moderate	Moderate	0.10	7.40	83.97	8.54
Moderate	Moderate	High	0.10	0.46	59.44	40.00
Moderate	High	Zero	0.19	41.29	58.06	0.45
Moderate	High	Low	0.10	8.40	83.58	7.92
Moderate	High	Moderate	0.10	0.52	59.44	39.94
Moderate	High	High	0.10	0.11	17.06	82.73
High	Zero	Zero	50.98	48.69	0.23	0.10
High	Zero	Low	11.24	84.13	4.53	0.10
High	Zero	Moderate	0.88	67.27	31.73	0.13
High	Zero	High	0.11	22.90	75.24	1.75
High	Low	Zero	11.97	82.79	5.14	0.10
High	Low	Low	0.74	65.09	34.03	0.14
High	Low	Moderate	0.11	23.85	74.65	1.39
High	Low	High	0.10	2.32	77.75	19.84
High	Moderate	Zero	1.07	67.60	31.20	0.13
High	Moderate	Low	0.11	22.75	75.47	1.67
High	Moderate	Moderate	0.10	2.78	79.06	18.07
High	Moderate	High	0.10	0.18	43.55	56.17
High	High	Zero	0.11	23.68	74.76	1.45
High	High	Low	0.10	2.92	79.44	17.54
High	High	Moderate	0.10	0.18	41.76	57.96
High	High	High	0.11	0.11	8.45	91.33

Table 4.13: Conditional probability table for fishery as an endpoint node.

		Fishery				
Biodiversity habitat	Productivity	Community	Zero	Low	Moderate	High
Zero	Zero	Zero	87.73	12.03	0.12	0.12
Zero	Zero	Low	68.29	31.48	0.13	0.10
Zero	Zero	Moderate	30.81	68.19	0.91	0.10
Zero	Zero	High	7.55	85.69	6.65	0.10
Zero	Low	Zero	67.24	32.53	0.13	0.10
Zero	Low	Low	31.19	67.81	0.90	0.10
Zero	Low	Moderate	7.57	84.23	8.10	0.10
Zero	Low	High	0.83	66.93	32.12	0.13
Zero	Moderate	Zero	34.88	64.32	0.71	0.10
Zero	Moderate	Low	7.61	84.75	7.55	0.10
Zero	Moderate	Moderate	0.80	68.37	30.70	0.14
Zero	Moderate	High	0.13	29.77	69.36	0.74
Zero	High	Zero	7.72	85.58	6.60	0.10
Zero	High	Low	0.92	66.59	32.36	0.12
Zero	High	Moderate	0.13	30.88	68.14	0.86
Zero	High	High	0.10	7.81	85.31	6.77
Low	Zero	Zero	70.20	29.56	0.14	0.10
Low	Zero	Low	32.31	66.87	0.73	0.10
Low	Zero	Moderate	6.71	86.00	7.19	0.10
Low	Zero	High	0.84	69.03	29.99	0.13
Low	Low	Zero	30.41	68.72	0.77	0.10
Low	Low	Low	8.14	83.89	7.87	0.10
Low	Low	Moderate	0.80	68.45	30.62	0.13
Low	Low	High	0.13	32.69	66.23	0.96
Low	Moderate	Zero	7.33	85.54	7.04	0.10
Low	Moderate	Low	0.86	67.55	31.46	0.13
Low	Moderate	Moderate	0.14	32.60	66.46	0.79
Low	Moderate	High	0.10	7.48	84.36	8.06
Low	High	Zero	0.97	68.71	30.20	0.13
Low	High	Low	0.13	31.69	67.16	1.02
Low	High	Moderate	0.10	7.35	84.54	8.02
Low	High	High	0.10	0.78	66.94	32.18
Moderate	Zero	Zero	33.00	66.15	0.75	0.10
Moderate	Zero	Low	8.34	84.08	7.48	0.10
Moderate	Zero	Moderate	0.85	67.79	31.22	0.14
Moderate	Zero	High	0.13	30.90	68.03	0.94
Moderate	Low	Zero	8.38	84.32	7.19	0.10
Moderate	Low	Low	0.68	65.96	33.24	0.13
Moderate	Low	Moderate	0.13	32.41	66.57	0.89
Moderate	Low	High	0.10	7.38	84.72	7.80
Moderate	Moderate	Zero	0.99	67.87	31.01	0.13
Moderate	Moderate	Low	0.13	31.78	67.30	0.79
Moderate	Moderate	Moderate	0.10	6.55	85.75	7.60
Moderate	Moderate	High	0.10	0.87	68.65	30.38
Moderate	High	Zero	0.13	32.22	66.84	0.81
Moderate	High	Low	0.10	6.75	86.10	7.05
Moderate	High	Moderate	0.10	0.79	68.62	30.50
Moderate	High	High	0.10	0.12	31.37	68.41
High	Zero	Zero	7.29	85.51	7.10	0.10
High	Zero	Low	0.74	68.76	30.37	0.13
High	Zero	Moderate	0.13	29.56	69.45	0.85
High	Zero	High	0.10	7.76	84.41	7.73
High	Low	Zero	0.84	68.63	30.40	0.13
High	Low	Low	0.14	32.04	67.05	0.77
High	Low	Moderate	0.10	7.55	84.55	7.80
High	Low	High	0.10	0.73	68.50	30.68
High	Moderate	Zero	0.13	31.87	67.08	0.92
High	Moderate	Low	0.10	7.32	85.51	7.07
High	Moderate	Moderate	0.10	0.78	66.89	32.24
High	Moderate	High	0.10	0.13	31.03	68.74
High	High	Zero	0.10	7.08	86.18	6.64
High	High	Low	0.10	0.77	65.30	33.83
High	High	Moderate	0.097655	0.13276	29.9938	69.7758
High	High	High	0.106652	0.10705	7.89701	91.8893

Table 4.14: Conditional probability table for productivity as an endpoint node.

			Productivity			
Resident time	Nutrient load	Estuary type	Zero	Low	Moderate	High
Zero	Zero	Zero	91.78	8.01	0.11	0.11
Zero	Zero	Low	69.17	30.61	0.12	0.10
Zero	Zero	Moderate	30.01	69.06	0.83	0.10
Zero	Zero	High	6.67	85.21	8.02	0.10
Zero	Low	Zero	66.61	33.16	0.13	0.10
Zero	Low	Low	30.71	68.38	0.80	0.10
Zero	Low	Moderate	7.05	85.74	7.11	0.10
Zero	Low	High	0.77	67.26	31.84	0.13
Zero	Moderate	Zero	31.98	67.03	0.90	0.10
Zero	Moderate	Low	7.75	85.08	7.07	0.10
Zero	Moderate	Moderate	1.00	68.51	30.37	0.13
Zero	Moderate	High	0.13	28.92	70.04	0.91
Zero	High	Zero	7.36	84.83	7.71	0.10
Zero	High	Low	0.92	67.56	31.40	0.12
Zero	High	Moderate	0.13	30.27	68.80	0.80
Zero	High	High	0.10	7.82	84.86	7.22
Low	Zero	Zero	69.02	30.74	0.14	0.10
Low	Zero	Low	31.00	68.05	0.84	0.10
Low	Zero	Moderate	6.68	86.19	7.03	0.10
Low	Zero	High	0.84	65.17	33.87	0.12
Low	Low	Zero	32.64	66.38	0.88	0.10
Low	Low	Low	6.98	85.45	7.48	0.10
Low	Low	Moderate	0.88	66.53	32.45	0.13
Low	Low	High	0.13	31.90	67.21	0.76
Low	Moderate	Zero	7.11	85.38	7.41	0.10
Low	Moderate	Low	0.86	66.50	32.51	0.13
Low	Moderate	Moderate	0.13	31.21	67.72	0.94
Low	Moderate	High	0.10	7.13	84.93	7.84
Low	High	Zero	0.89	68.66	30.30	0.14
Low	High	Low	0.13	30.93	68.18	0.76
Low	High	Moderate	0.10	6.88	85.69	7.33
Low	High	High	0.10	0.77	67.05	32.08
Moderate	Zero	Zero	29.26	69.71	0.93	0.10
Moderate	Zero	Low	7.47	85.19	7.24	0.10
Moderate	Zero	Moderate	0.93	68.60	30.34	0.12
Moderate	Zero	High	0.13	30.73	68.23	0.91
Moderate	Low	Zero	7.00	84.80	8.10	0.10
Moderate	Low	Low	0.89	68.25	30.72	0.14
Moderate	Low	Moderate	0.14	30.56	68.41	0.89
Moderate	Low	High	0.10	6.70	85.41	7.79
Moderate	Moderate	Zero	0.75	68.25	30.88	0.13
Moderate	Moderate	Low	0.13	31.65	67.29	0.93
Moderate	Moderate	Moderate	0.10	6.46	85.76	7.67
Moderate	Moderate	High	0.10	0.78	67.18	31.94
Moderate	High	Zero	0.13	29.03	69.99	0.86
Moderate	High	Low	0.10	7.74	85.05	7.11
Moderate	High	Moderate	0.10	0.83	67.74	31.33
Moderate	High	High	0.10	0.14	30.65	69.11
High	Zero	Zero	7.37	84.23	8.31	0.10
High	Zero	Low	0.90	69.40	29.57	0.13
High	Zero	Moderate	0.12	32.97	66.10	0.81
High	Zero	High	0.10	7.48	85.08	7.34
High	Low	Zero	0.94	67.94	30.98	0.14
High	Low	Low	0.12	32.15	66.74	1.00
High	Low	Moderate	0.10	8.01	85.37	6.51
High	Low	High	0.10	0.83	68.76	30.30
High	Moderate	Zero	0.14	29.81	69.24	0.81
High	Moderate	Low	0.10	7.16	85.41	7.33
High	Moderate	Moderate	0.10	0.74	68.76	30.40
High	Moderate	High	0.10	0.14	31.26	68.50
High	High	Zero	0.10	7.68	84.94	7.27
High	High	Low	0.10	0.93	68.15	30.83
High	High	Moderate	0.10	0.13	29.05	70.72
High	High	High	0.11	0.11	8.04	91.75

4.8 Supplementary material (S2)

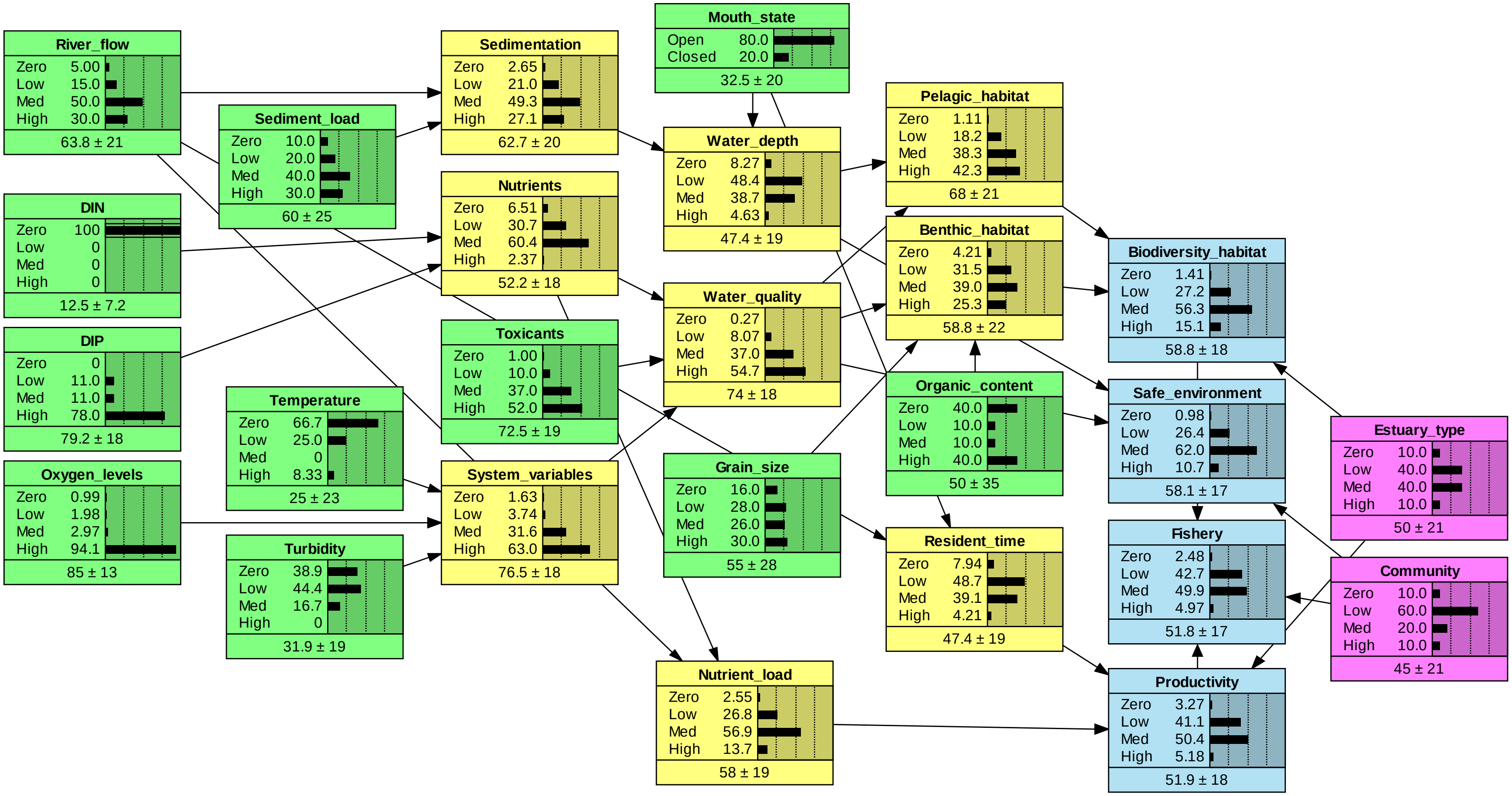

Fig. S1: Bayesian network relative risk model for Scenario 1 in uMvoti Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

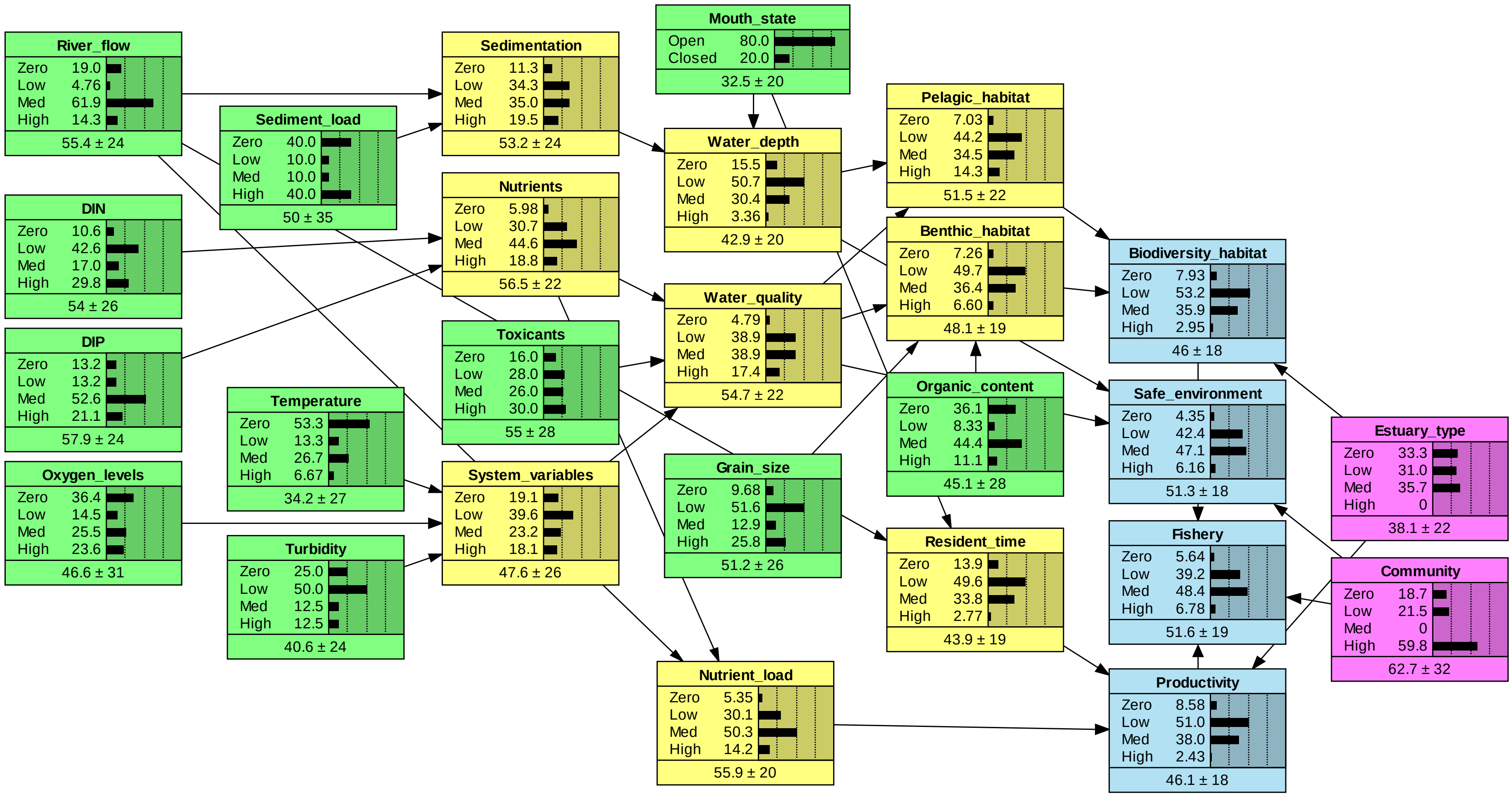

Fig. S2: Bayesian network relative risk model for Scenario 1 in uMvoti Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

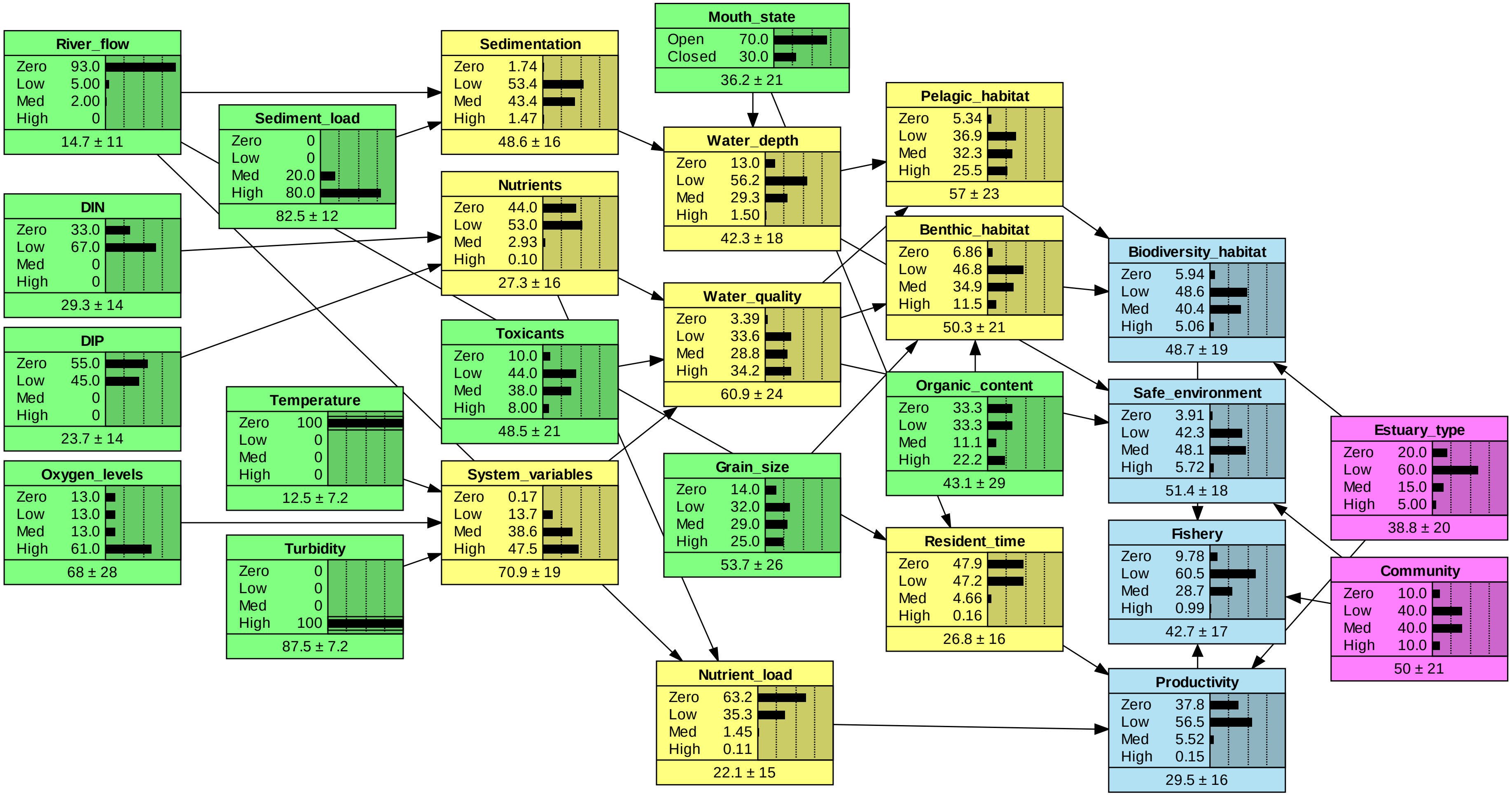

Fig. S3: Bayesian network relative risk model for Scenario 1 in Thukela Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

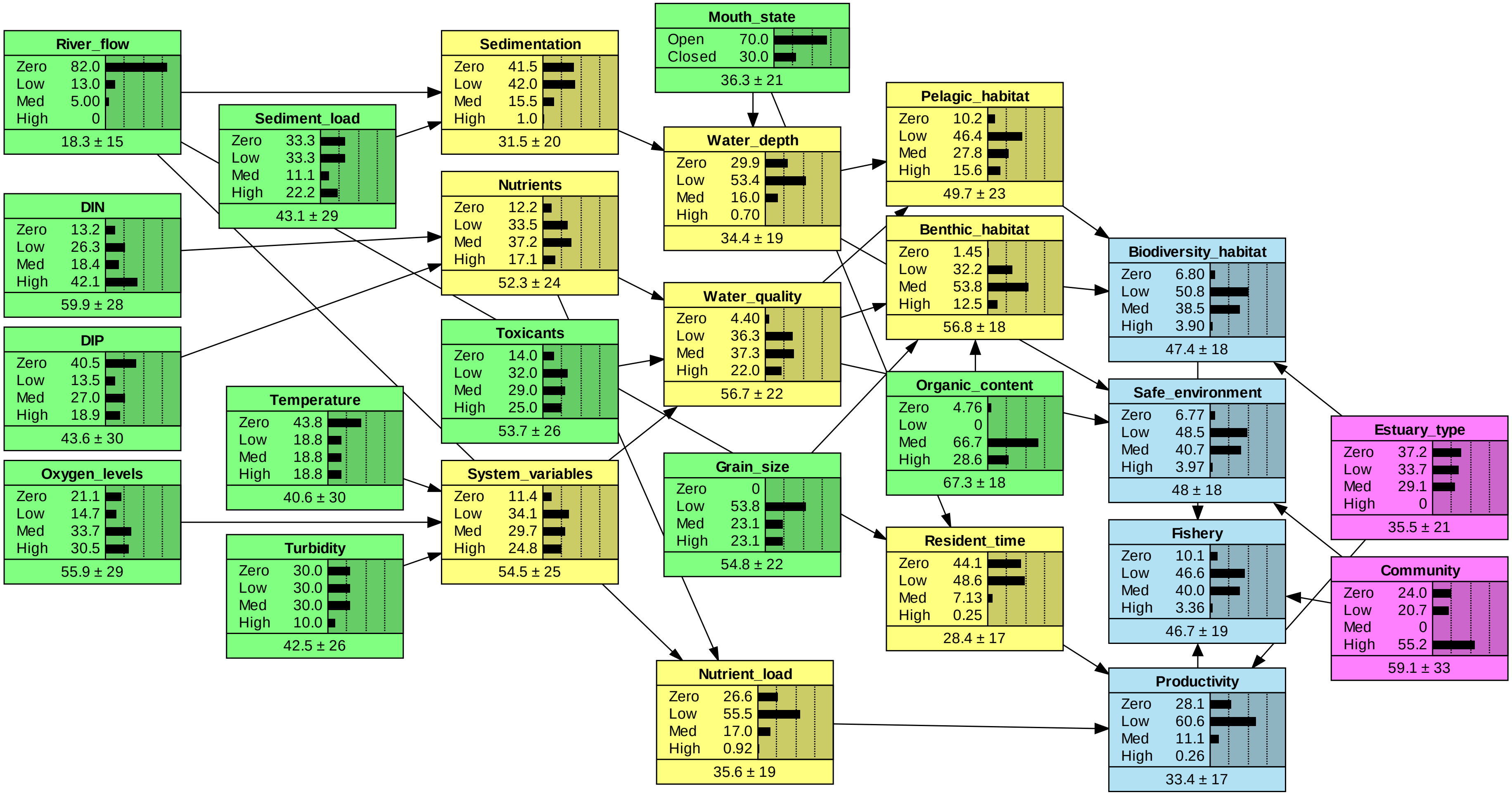

Fig. S4: Bayesian network relative risk model for Scenario 1 in Thukela Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

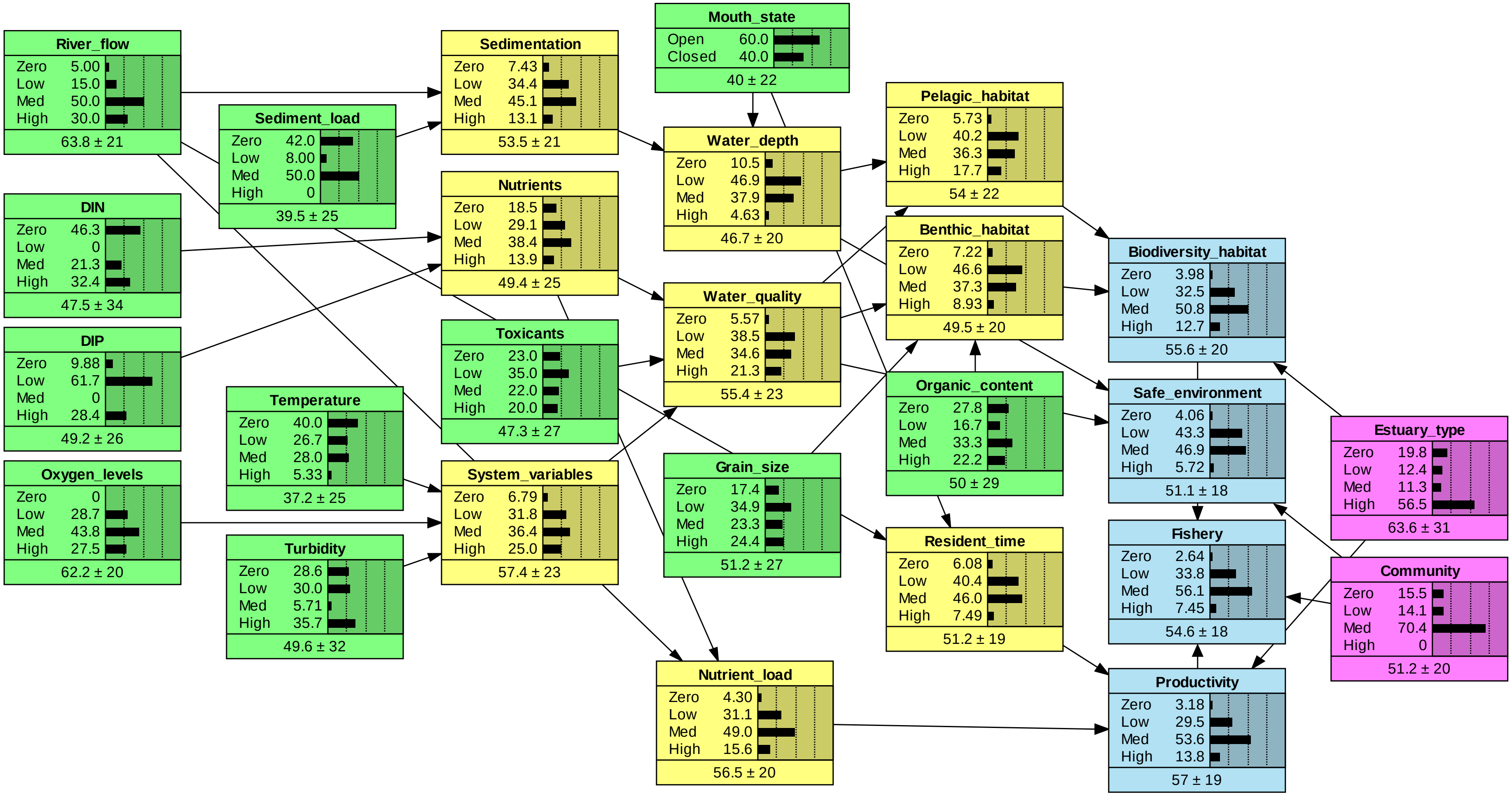

Fig. S5: Bayesian network relative risk model for Scenario 1 in aMatikulu/Nyoni Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

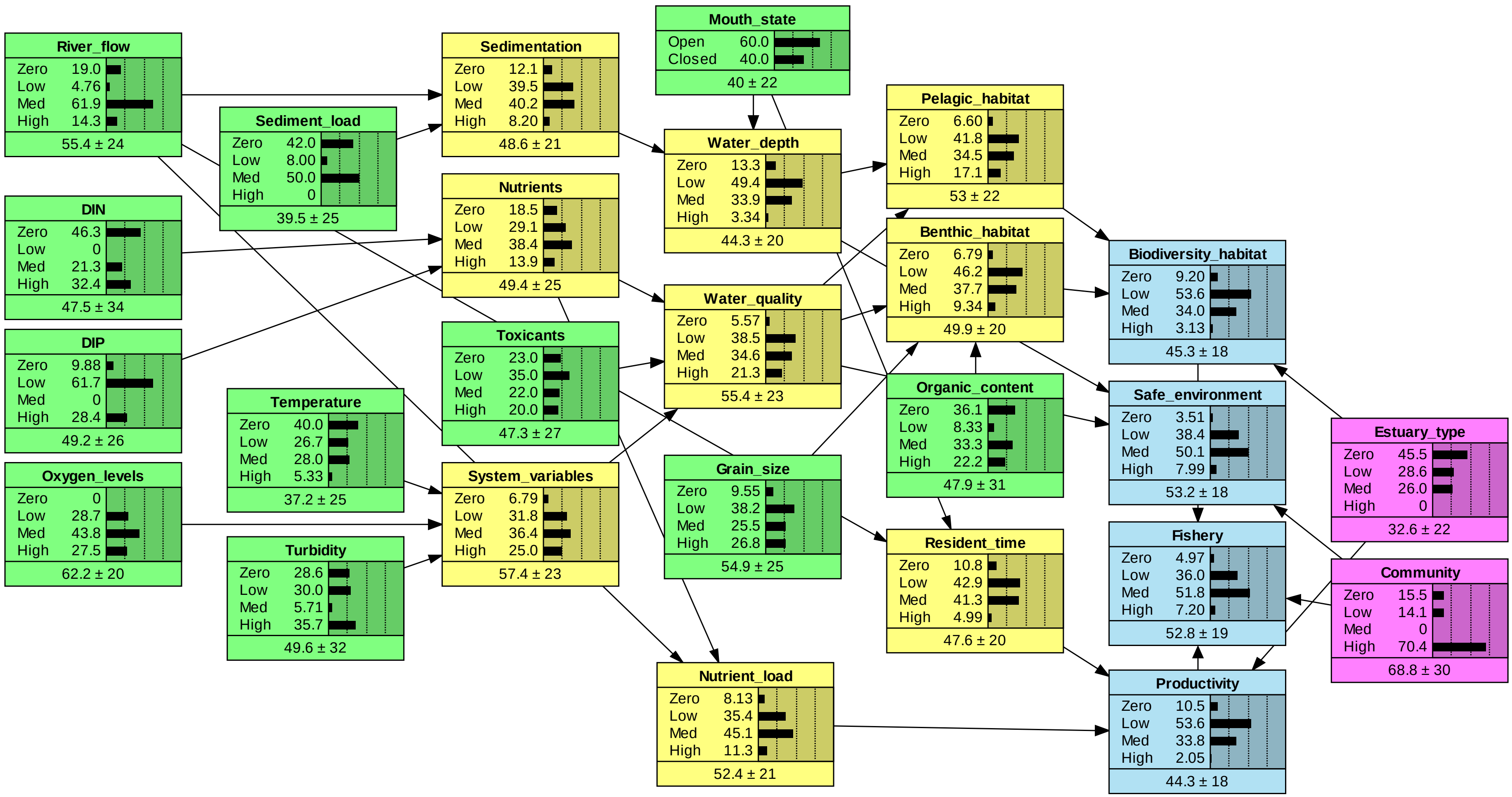

Fig. S6: Bayesian network relative risk model for Scenario 1 in aMatikulu/Nyoni Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

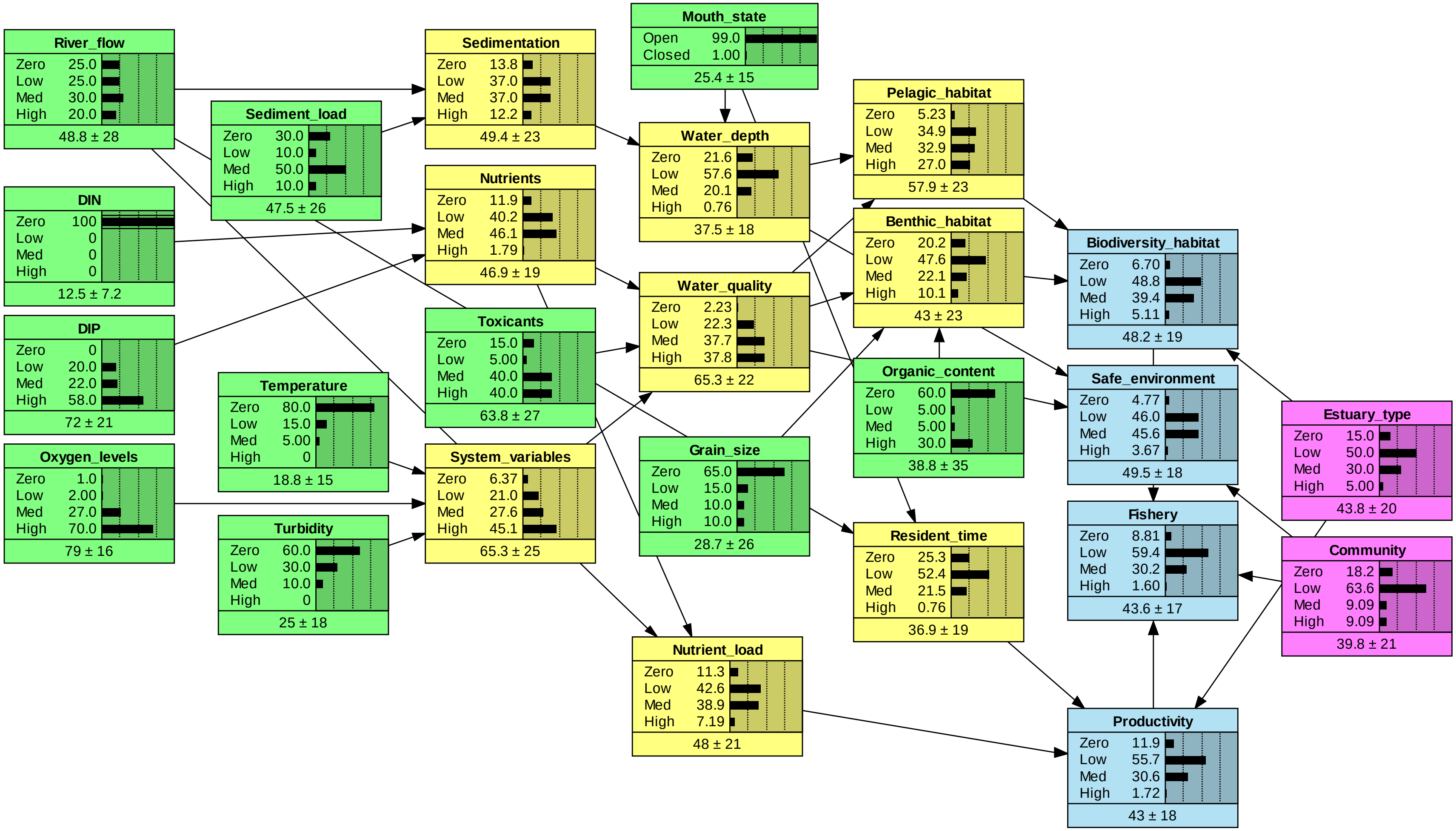

Fig. S7: Bayesian network relative risk model for Scenario 2 in uMvoti Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

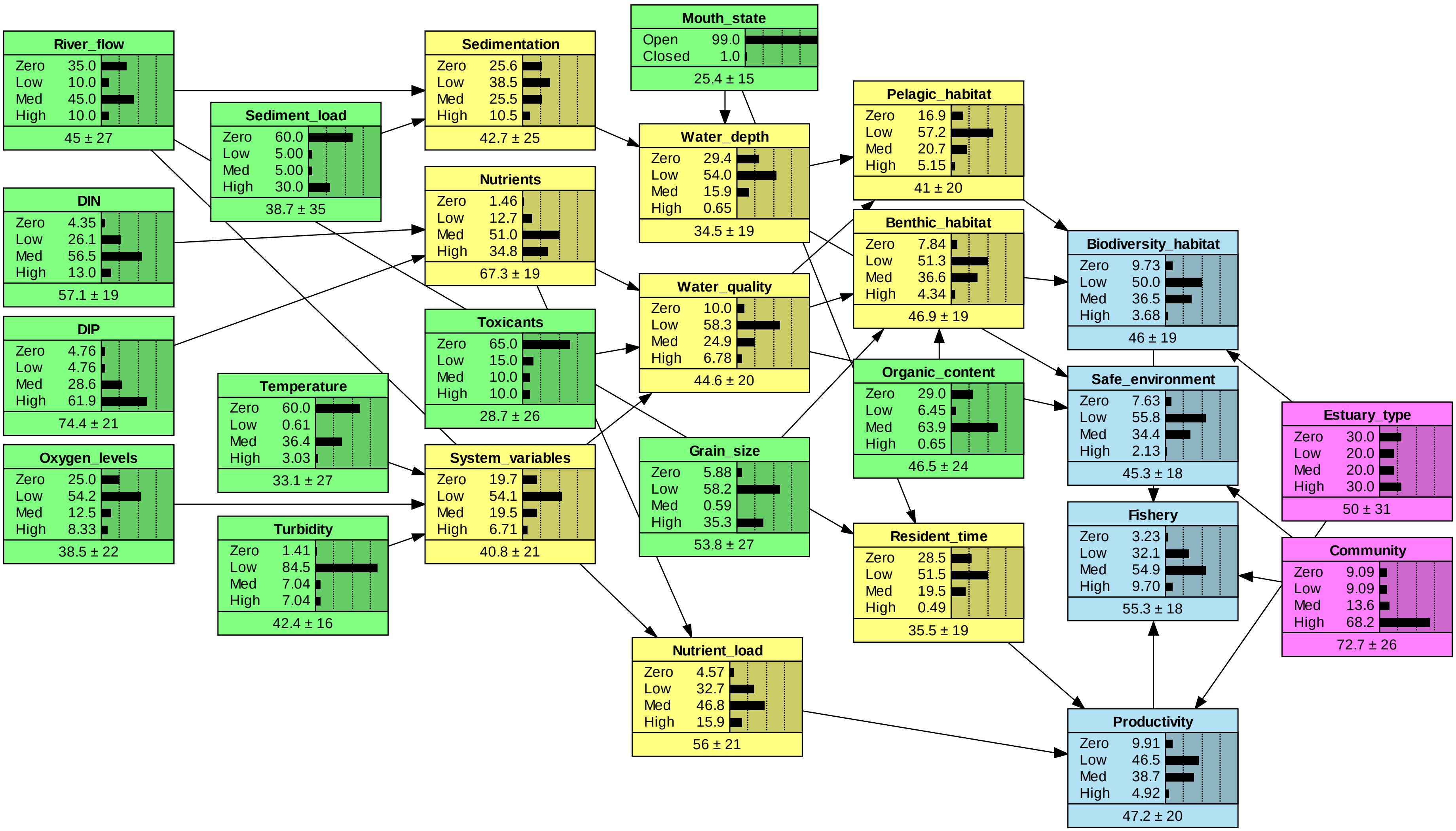

Fig. S8: Bayesian network relative risk model for Scenario 2 in uMvoti Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

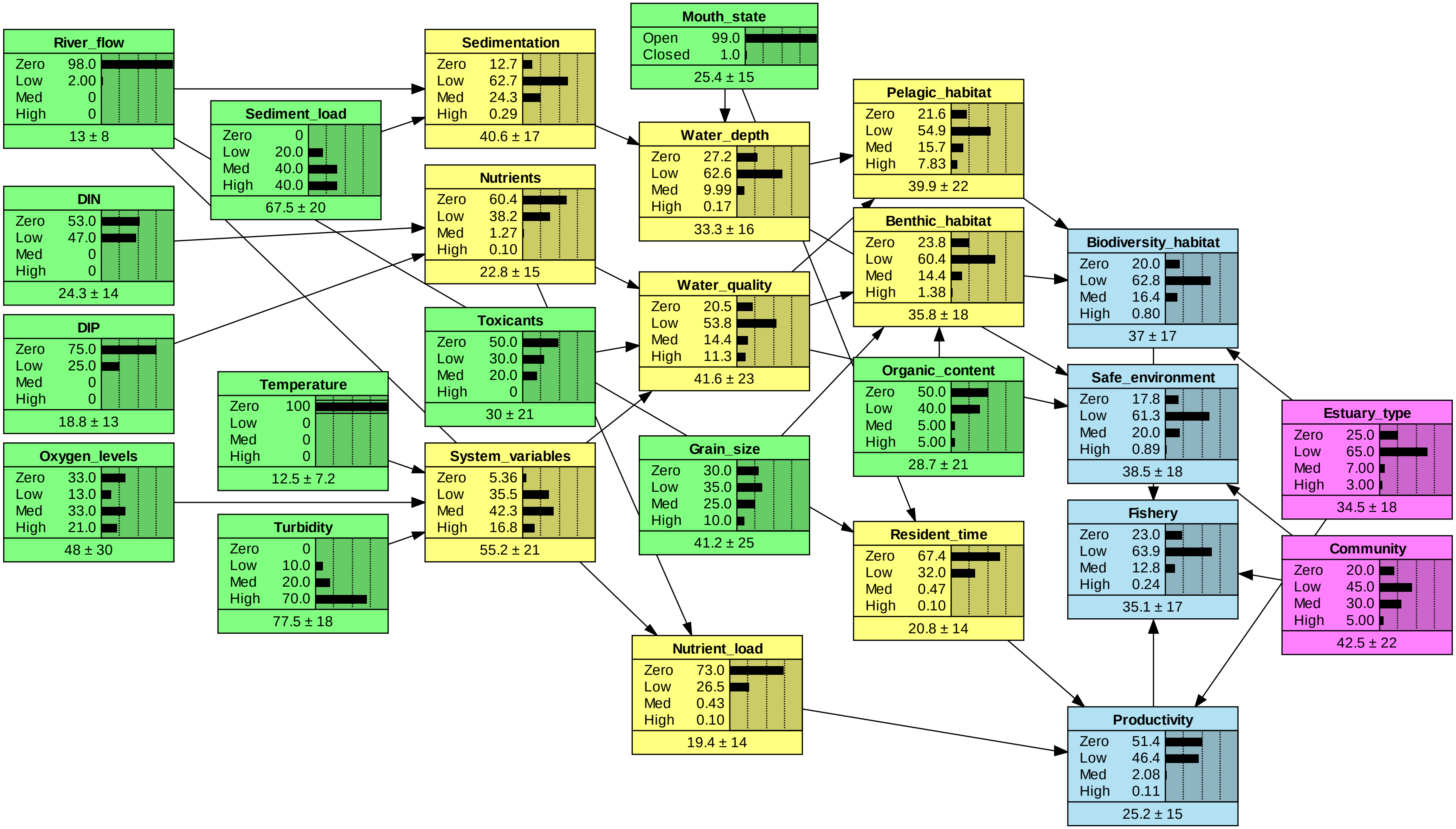

Fig S9: Bayesian network relative risk model for Scenario 2 in Thukela Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

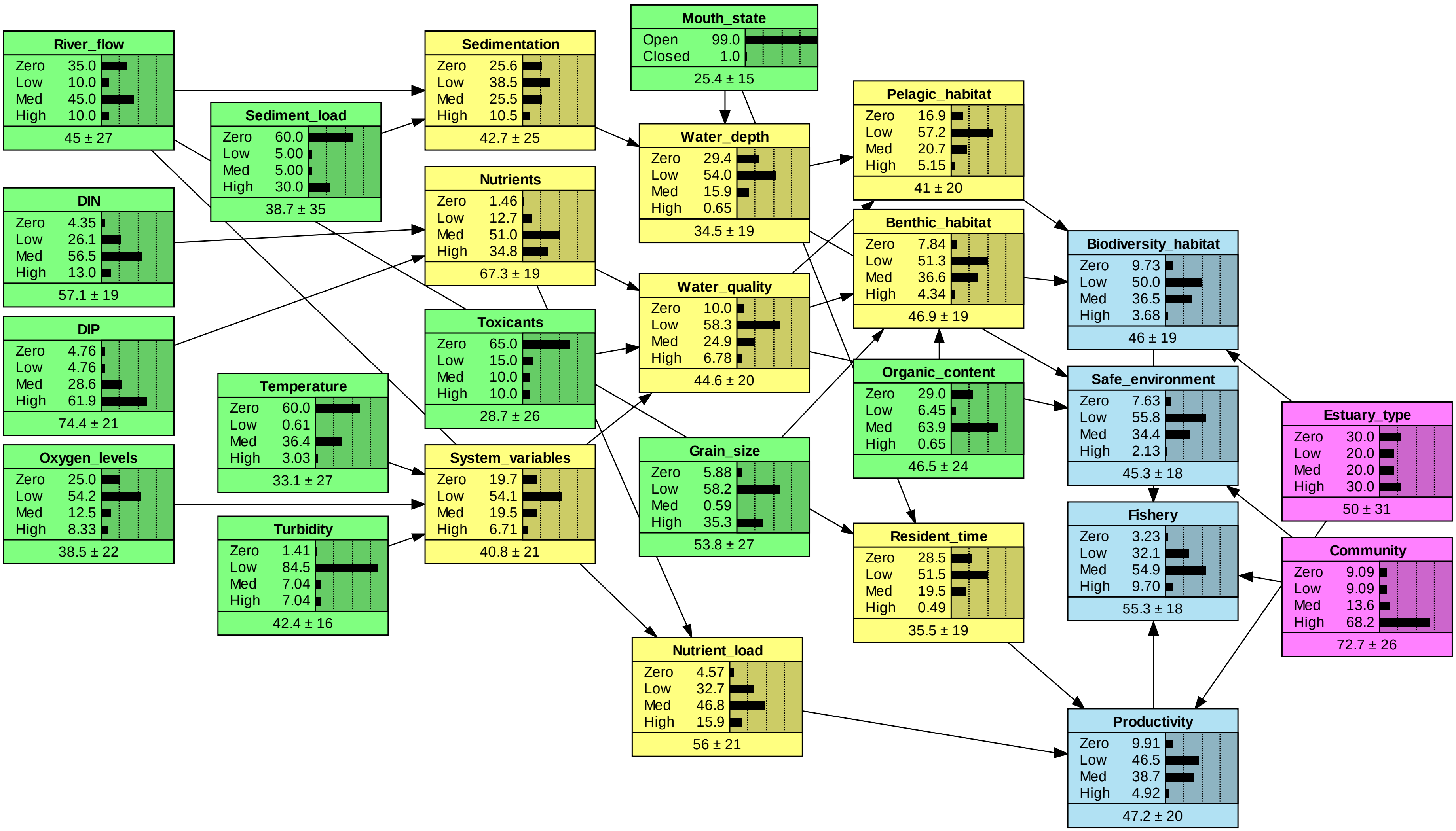

Fig. S10: Bayesian network relative risk model for Scenario 2 in Thukela Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

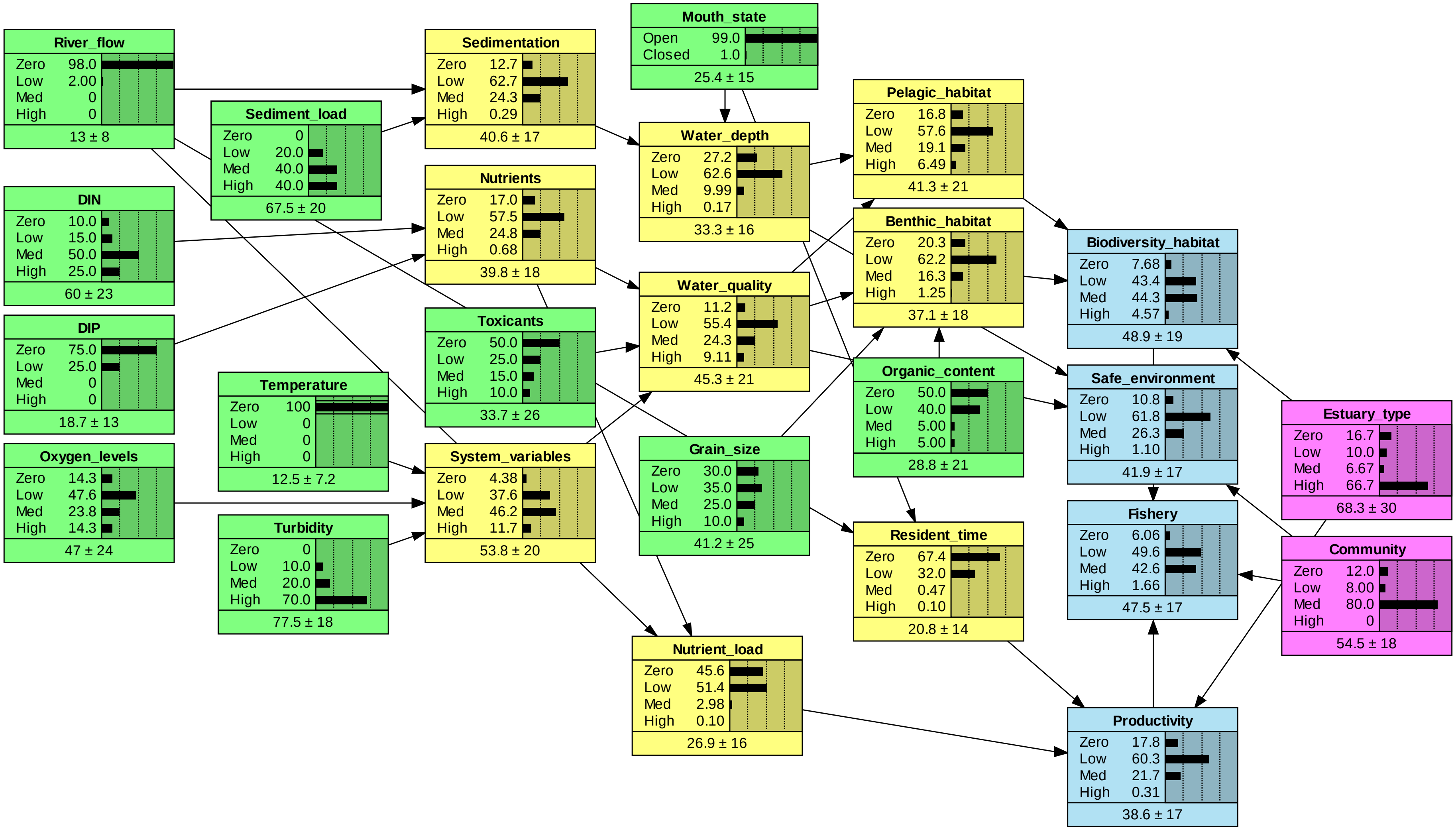

Fig. S11: Bayesian network relative risk model for Scenario 2 in aMatikulu/Nyoni Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

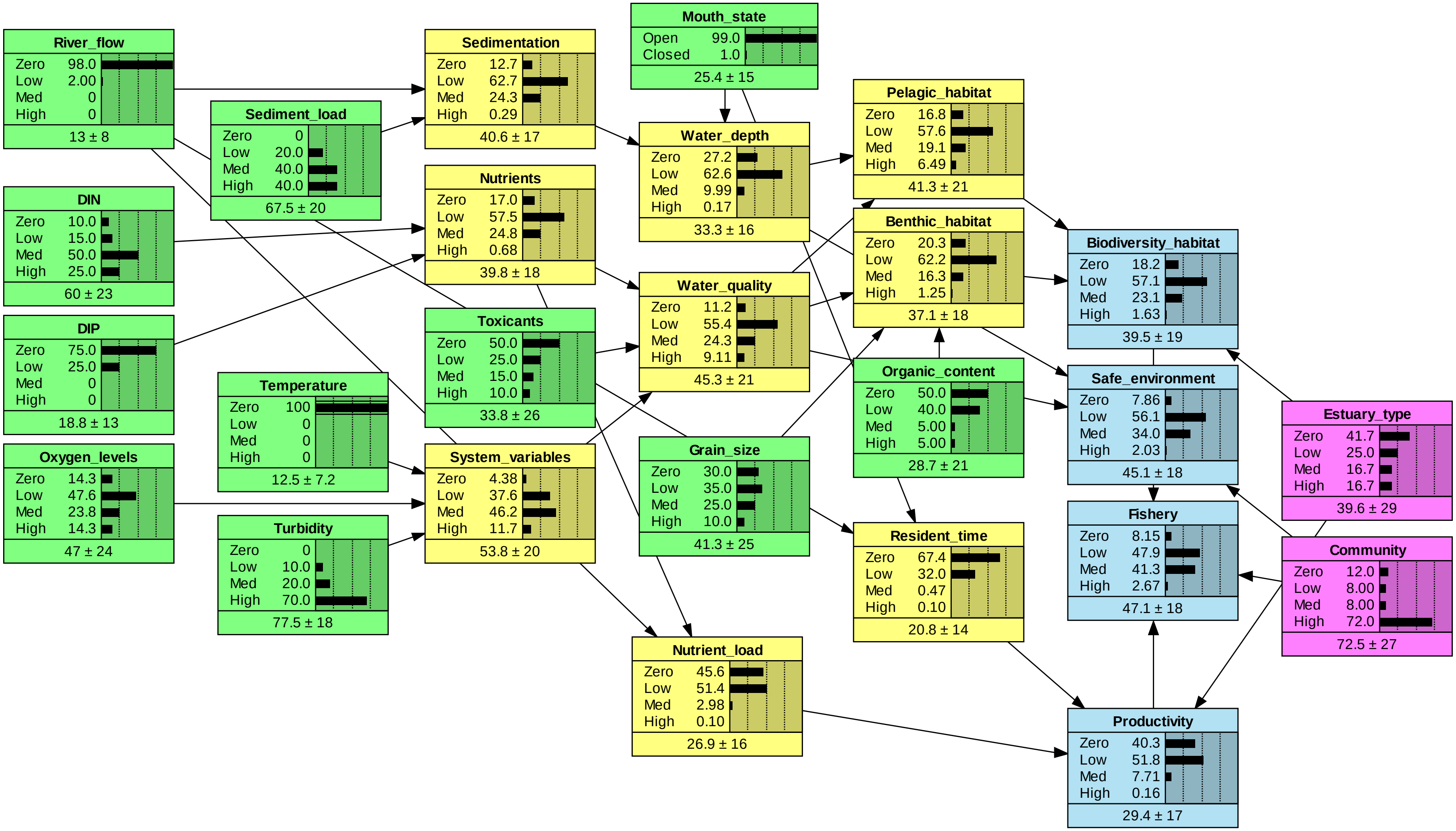

Fig. S12: Bayesian network relative risk model for Scenario 2 in aMatikulu/Nyoni Estuary (High flow flow period). Bottom values in each node presents mean risk score ±SD.

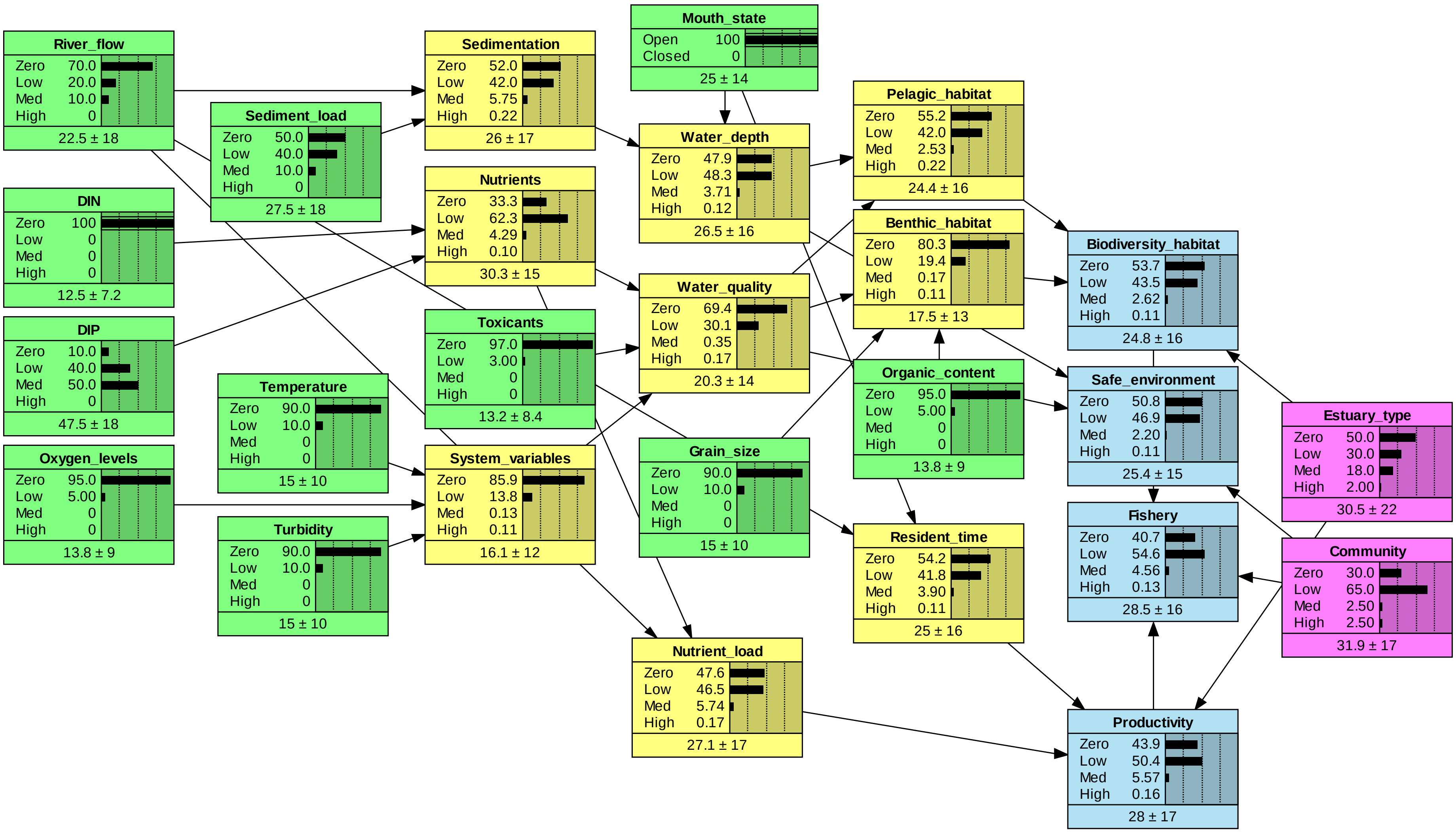

Fig. S13: Bayesian network relative risk model for Scenario 3 in uMvoti Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

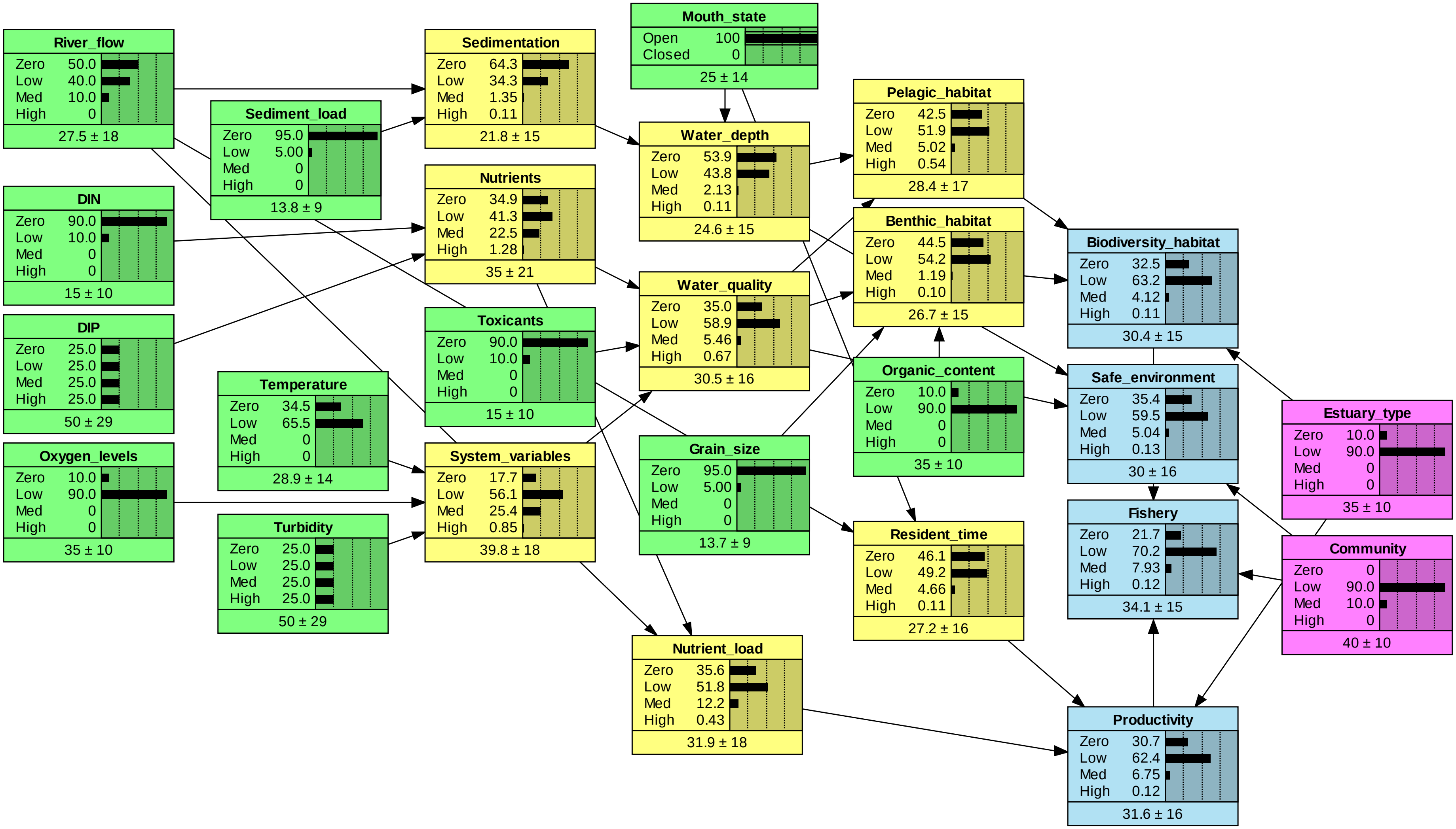

Fig. S14: Bayesian network relative risk model for Scenario 3 in uMvoti Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

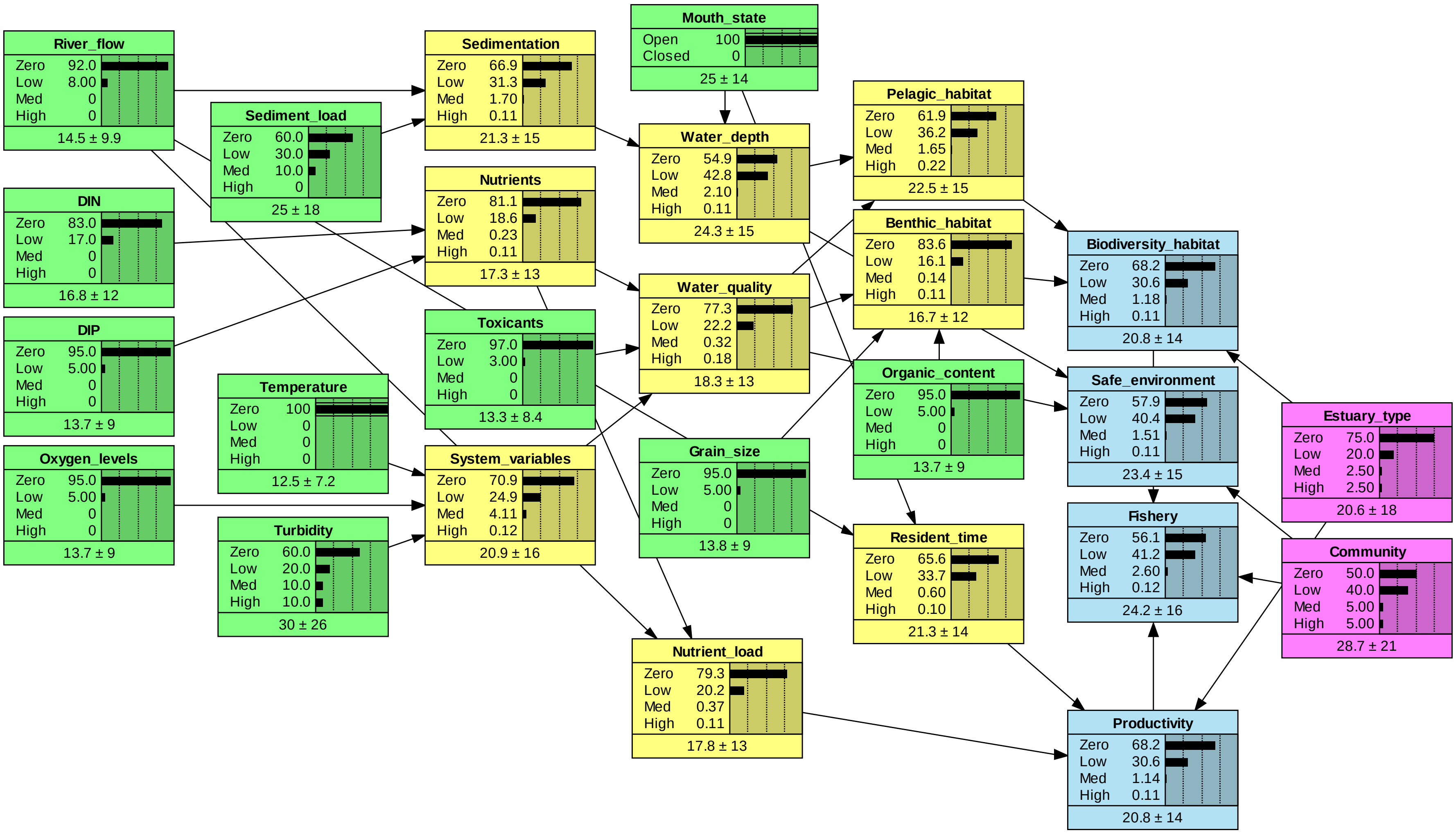

Fig. S15: Bayesian network relative risk model for Scenario 3 in Thukela Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

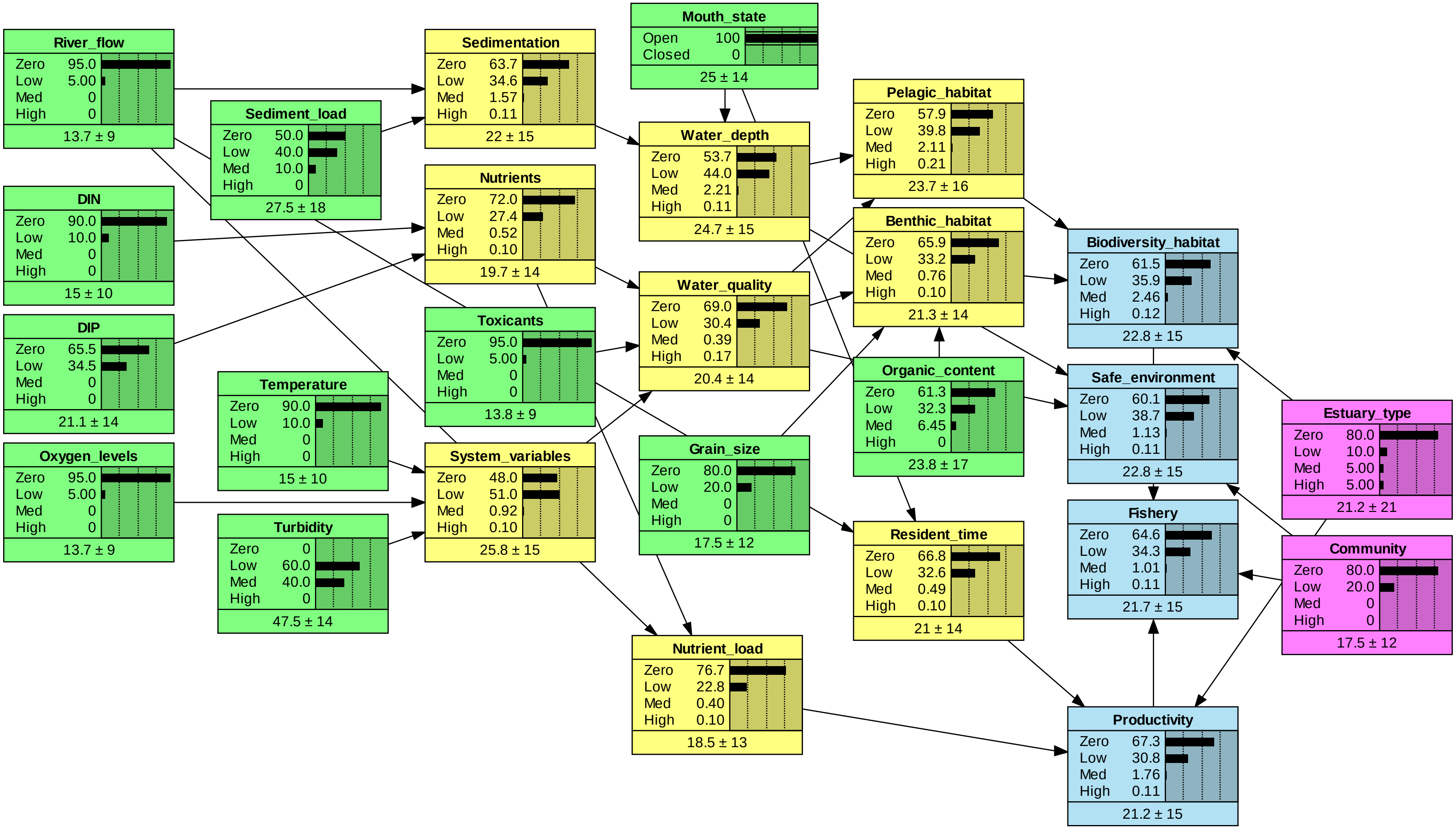

Fig. S16: Bayesian network relative risk model for Scenario 3 in Thukela Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

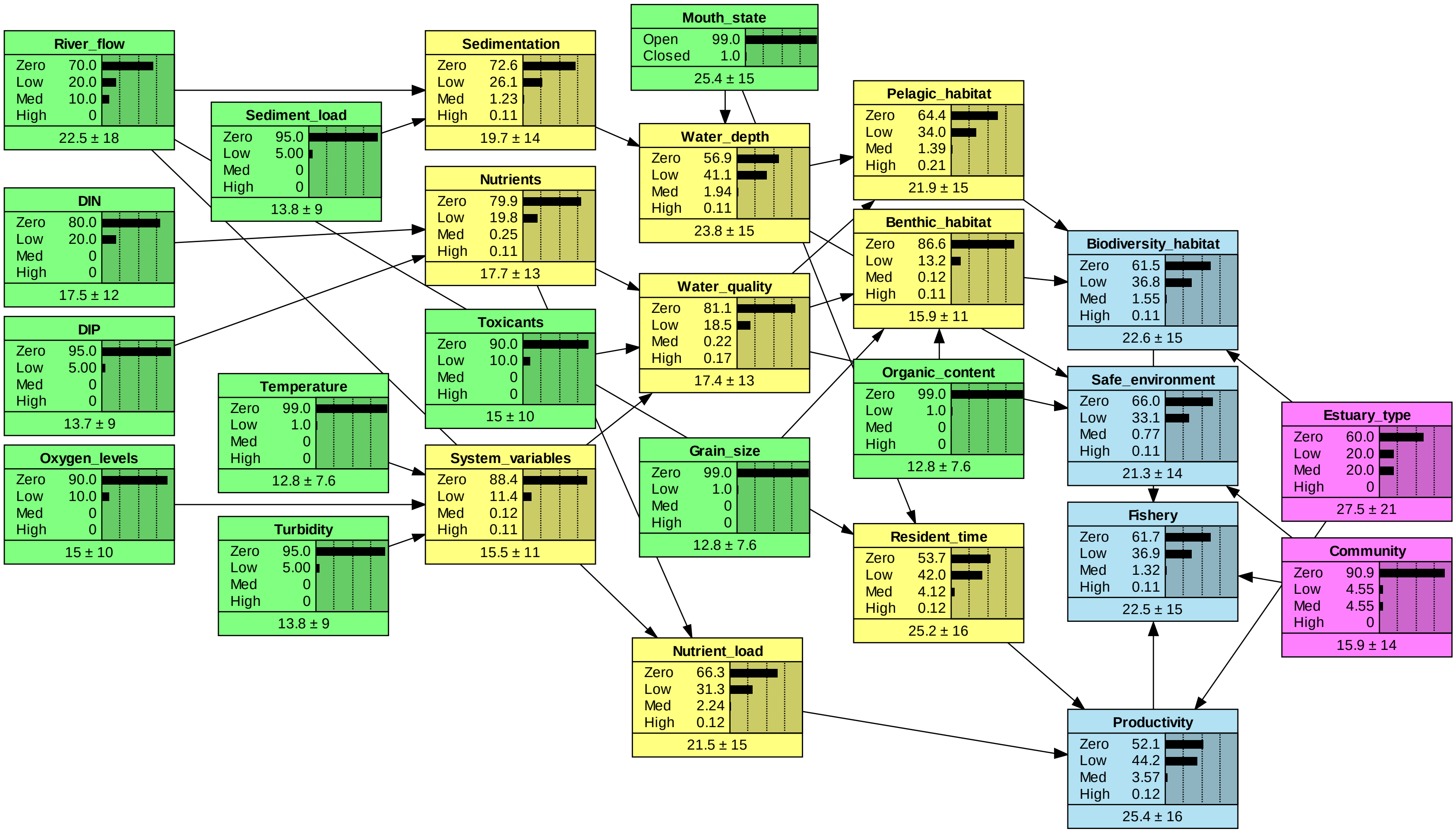

Fig. S17: Bayesian network relative risk model for Scenario 3 in aMatikulu/Nyoni Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

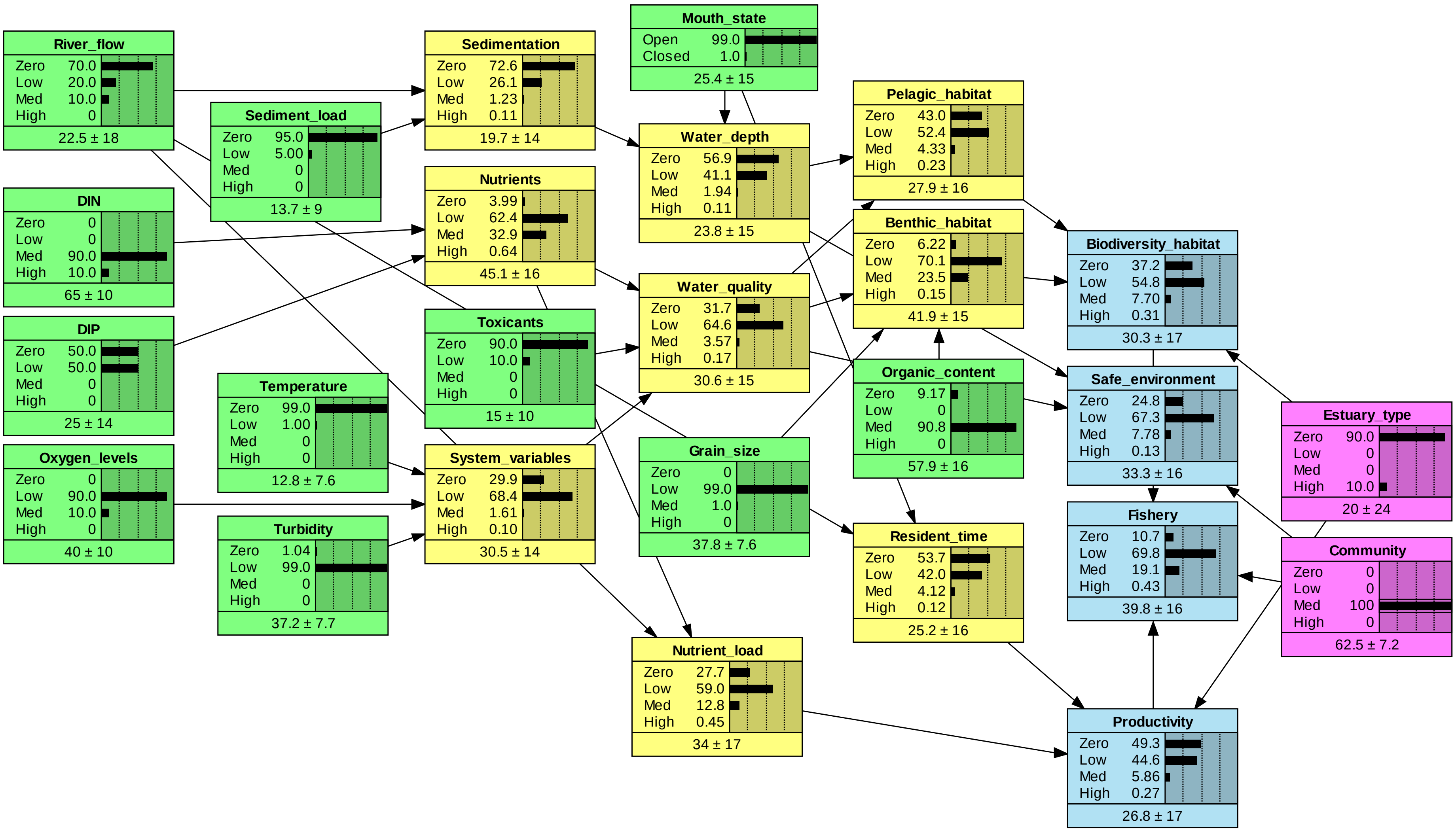

Fig. S18: Bayesian network relative risk model for Scenario 3 in aMatikulu/Nyoni Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

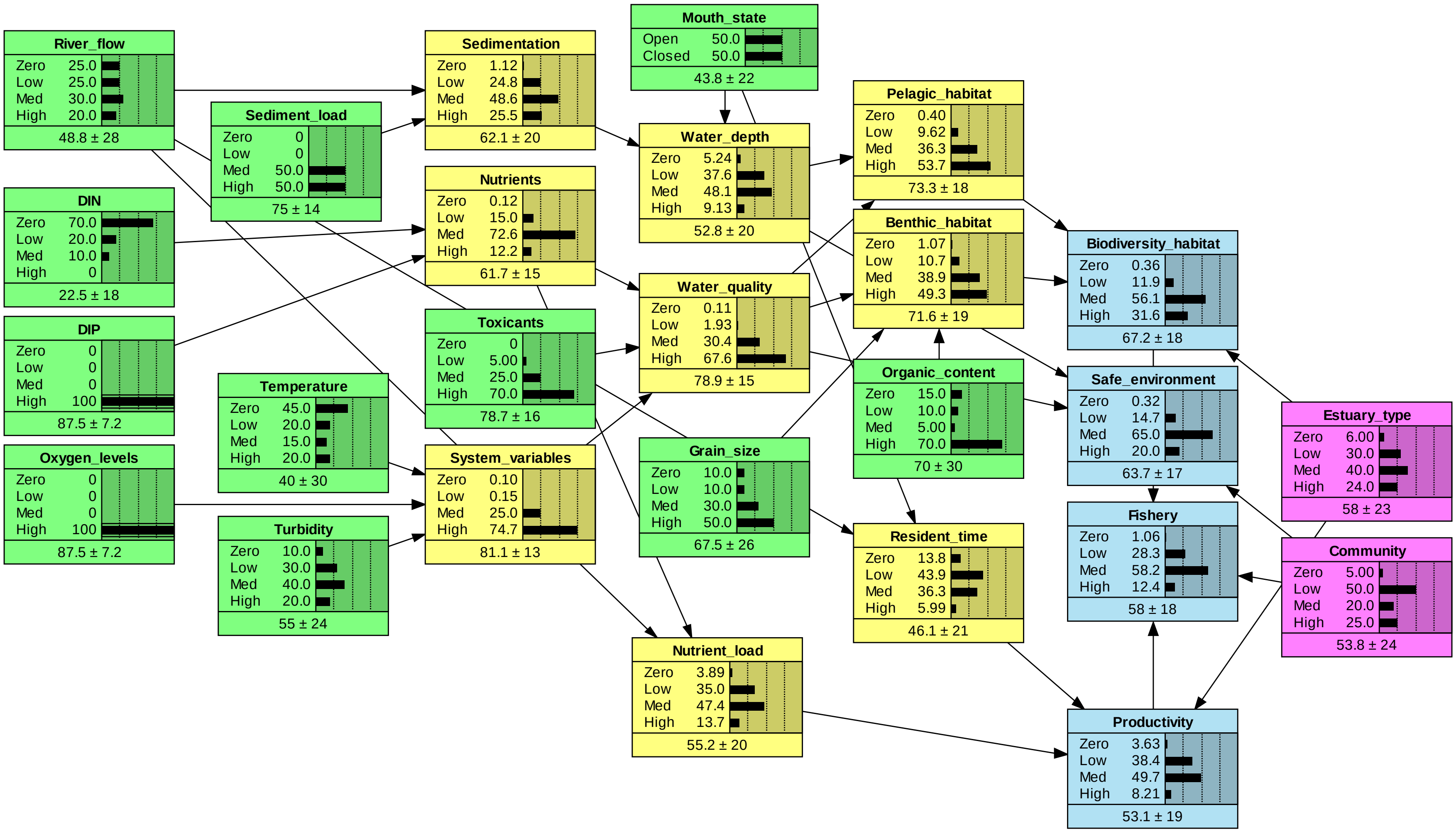

Fig. S19: Bayesian network relative risk model for Scenario 4 in uMvoti Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

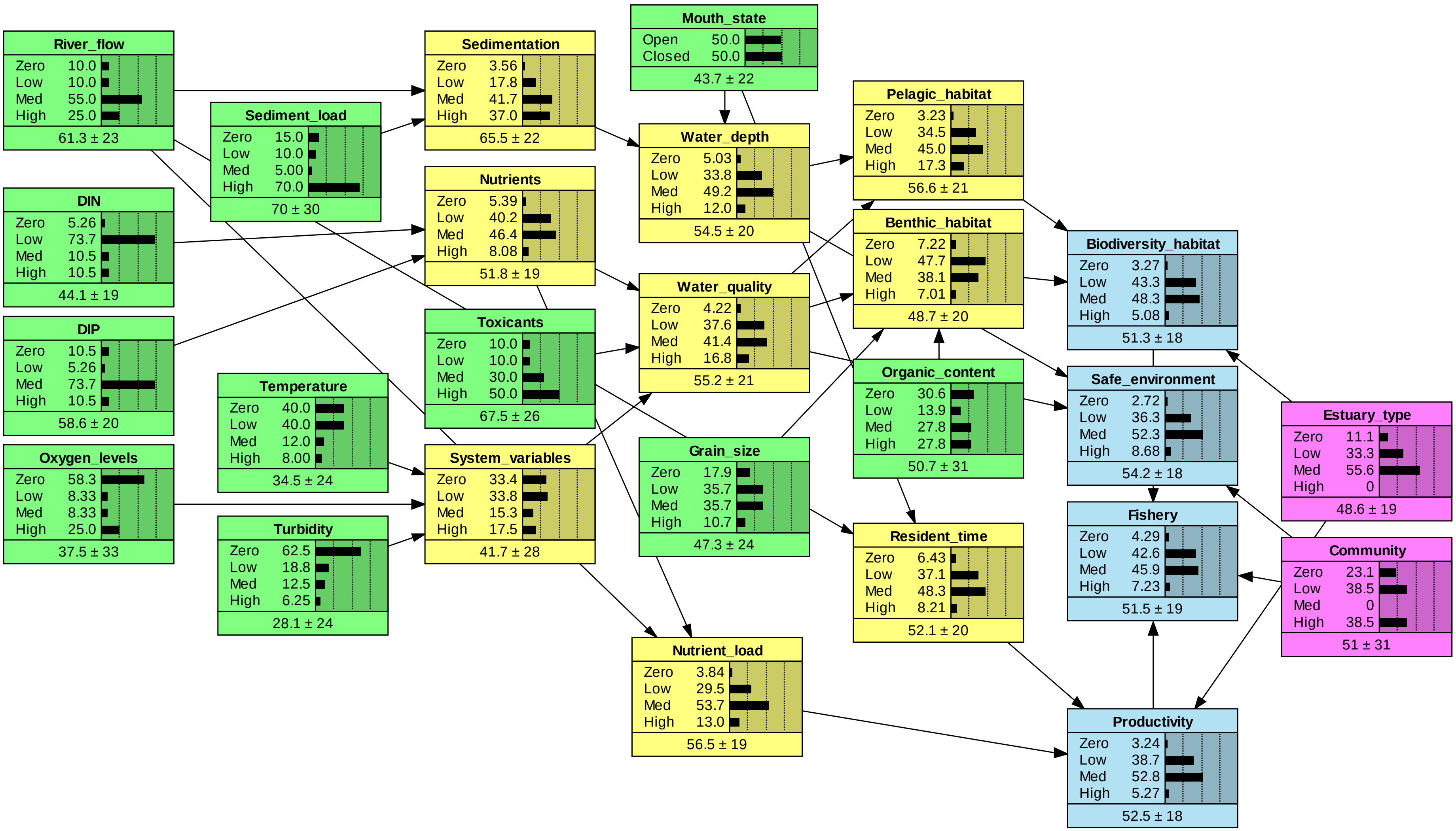

Fig S20: Bayesian network relative risk model for Scenario 4 in uMvoti Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

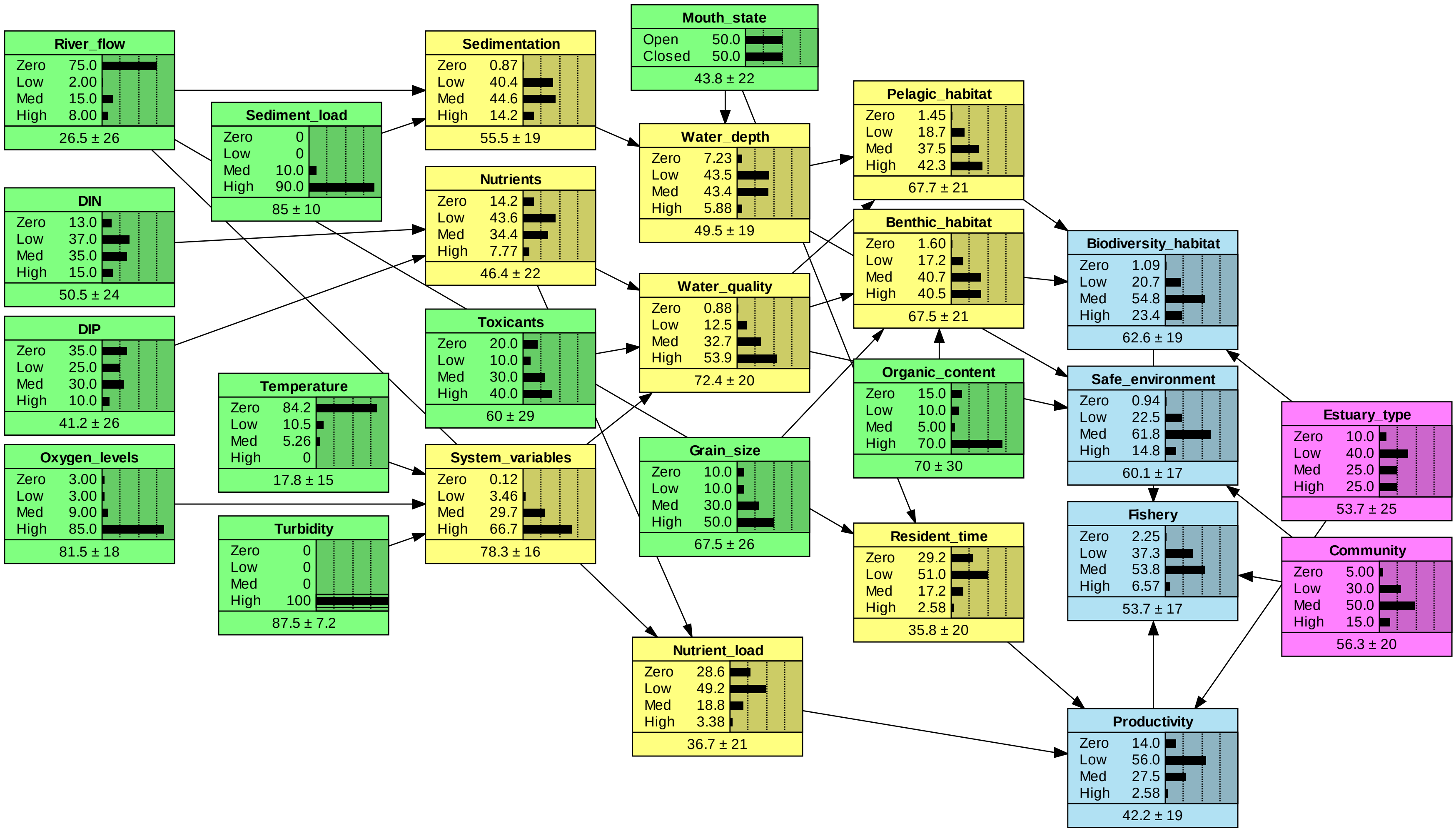

Fig. S21: Bayesian network relative risk model for Scenario 4 in Thukela Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

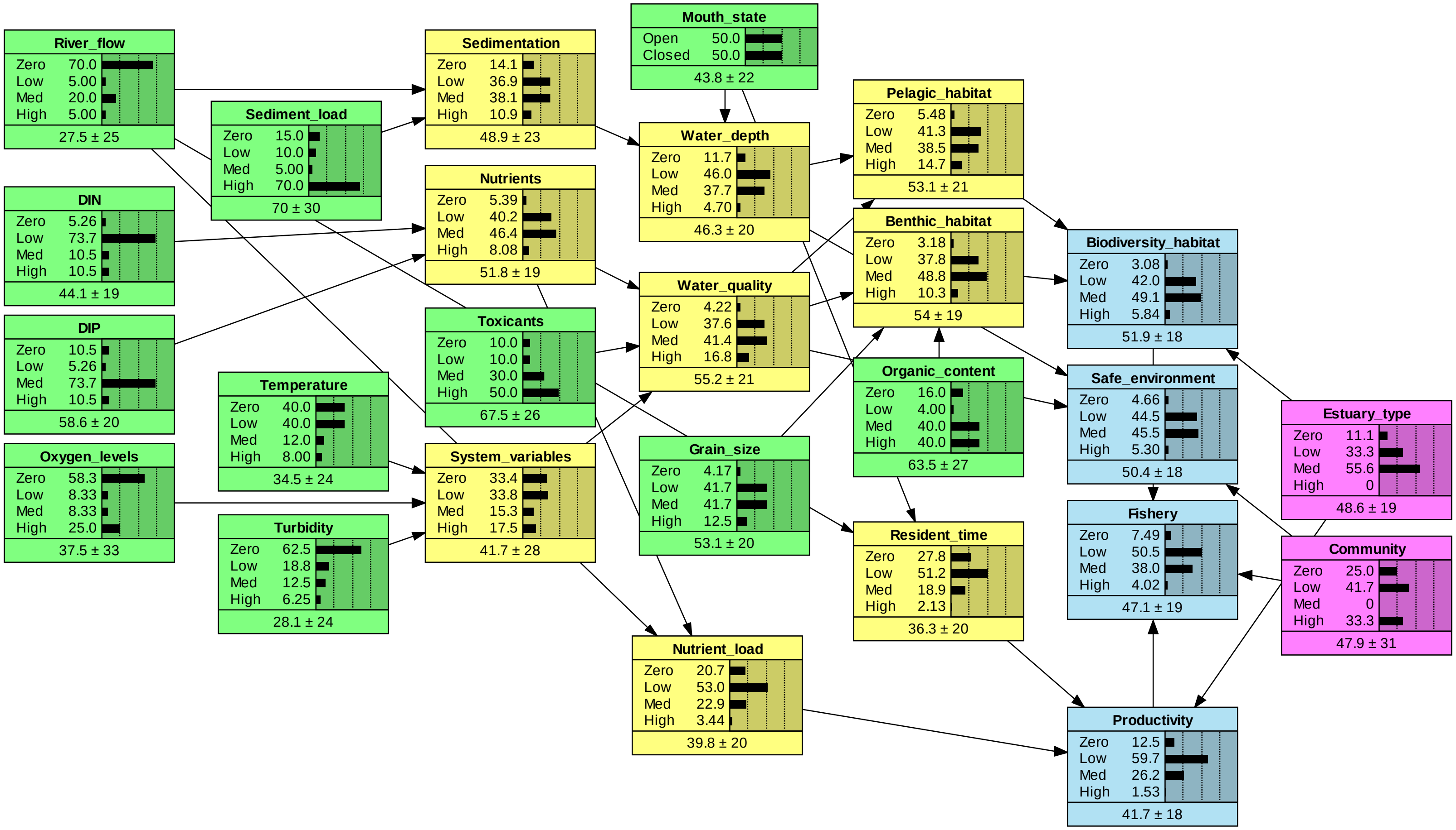

Fig. S22: Bayesian network relative risk model for Scenario 4 in Thukela Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

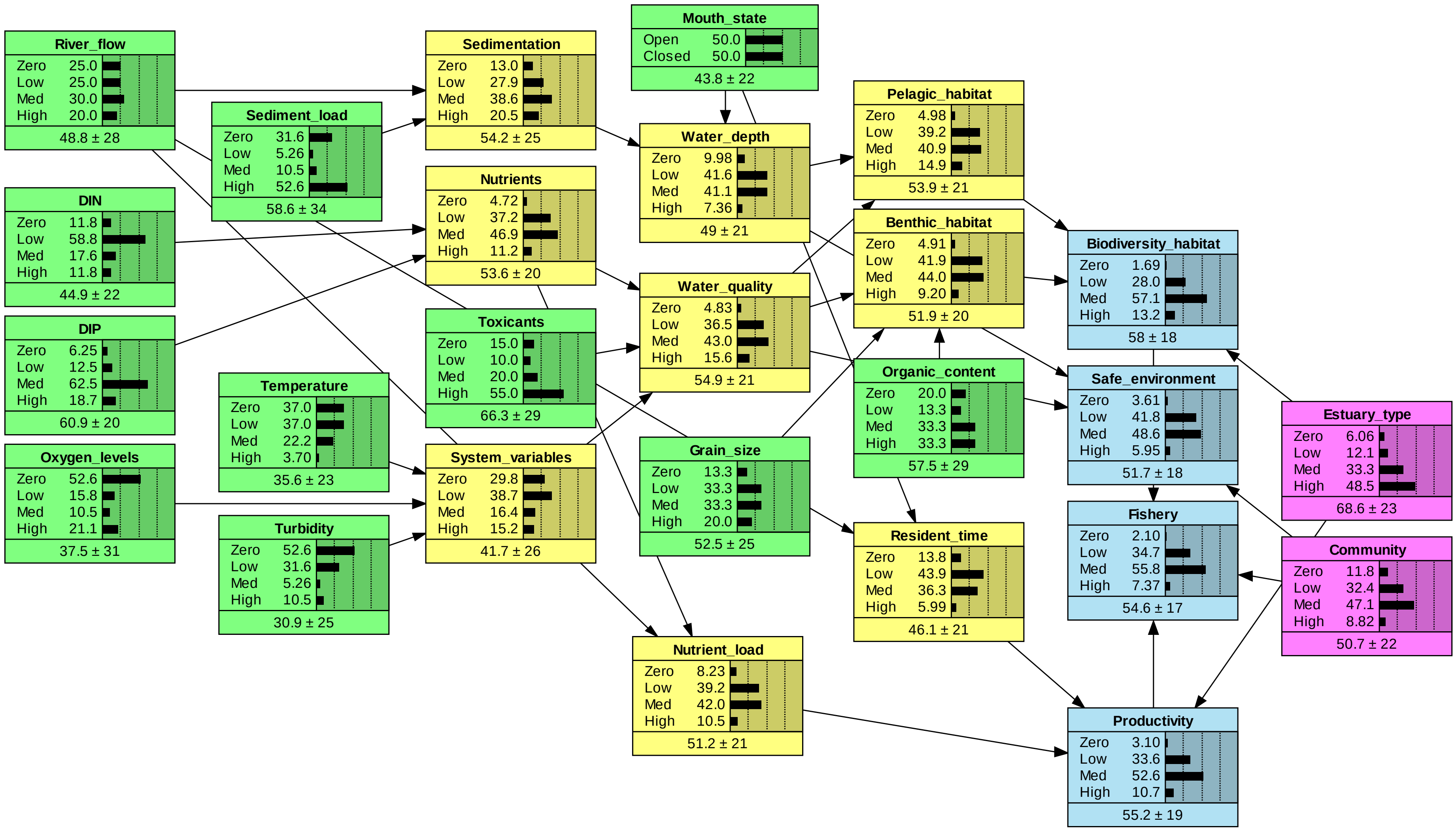

Fig. S23: Bayesian network relative risk model for Scenario 4 in aMatikulu/Nyoni Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

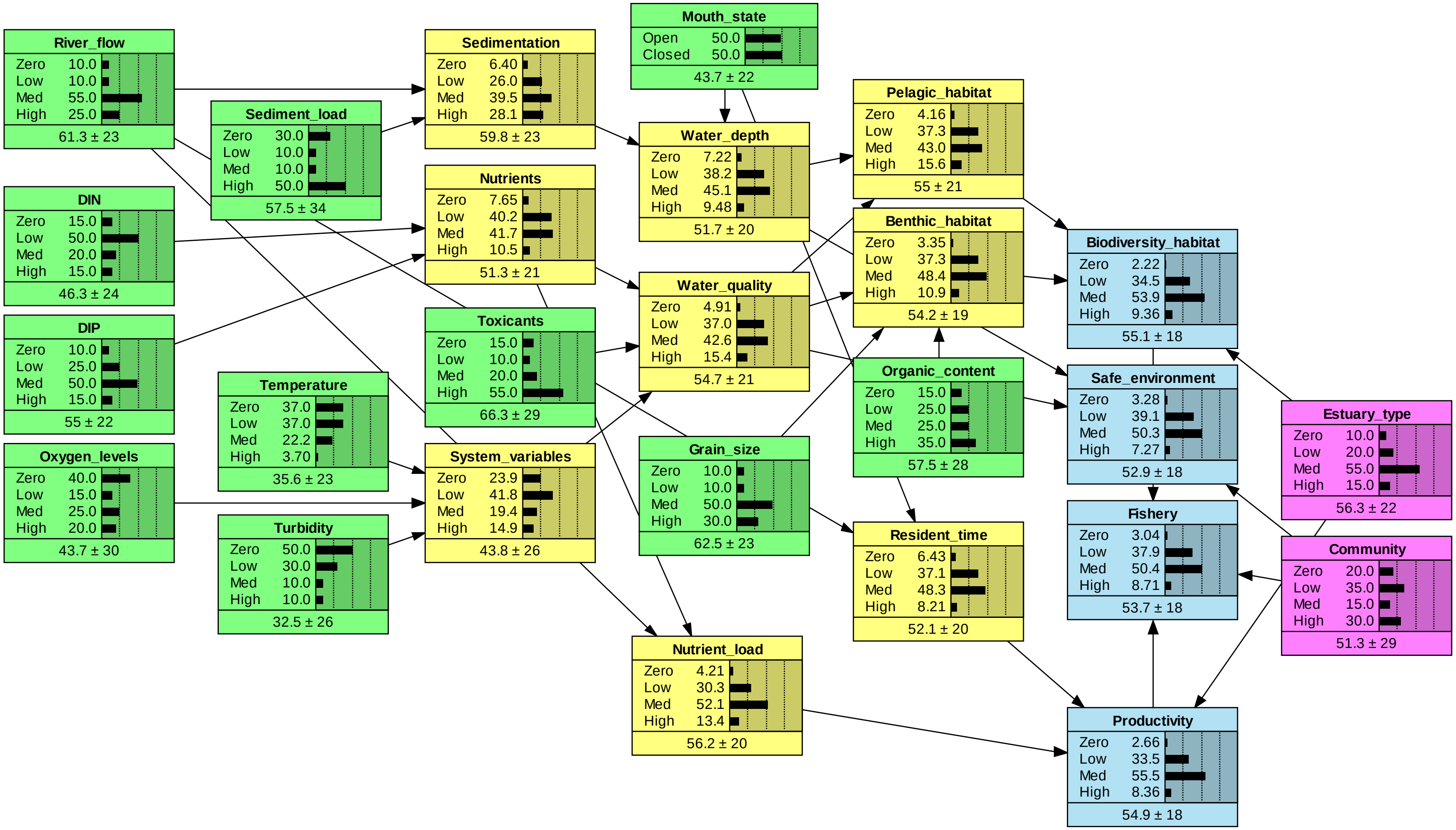

Fig. S24: Bayesian network relative risk model for Scenario 4 in aMatikulu/Nyoni Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

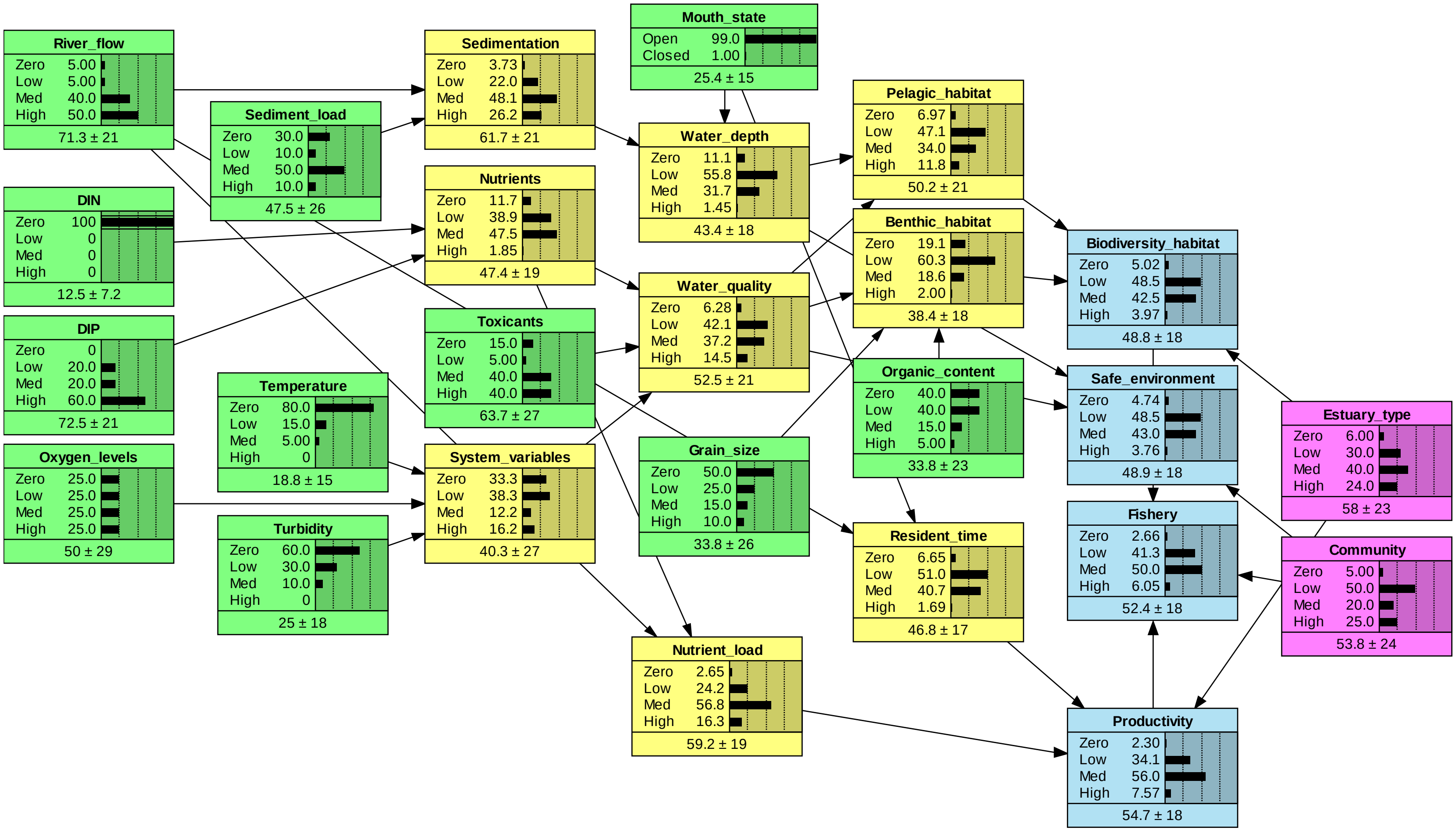

Fig. S25: Bayesian network relative risk model for Scenario 5 in uMvoti Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

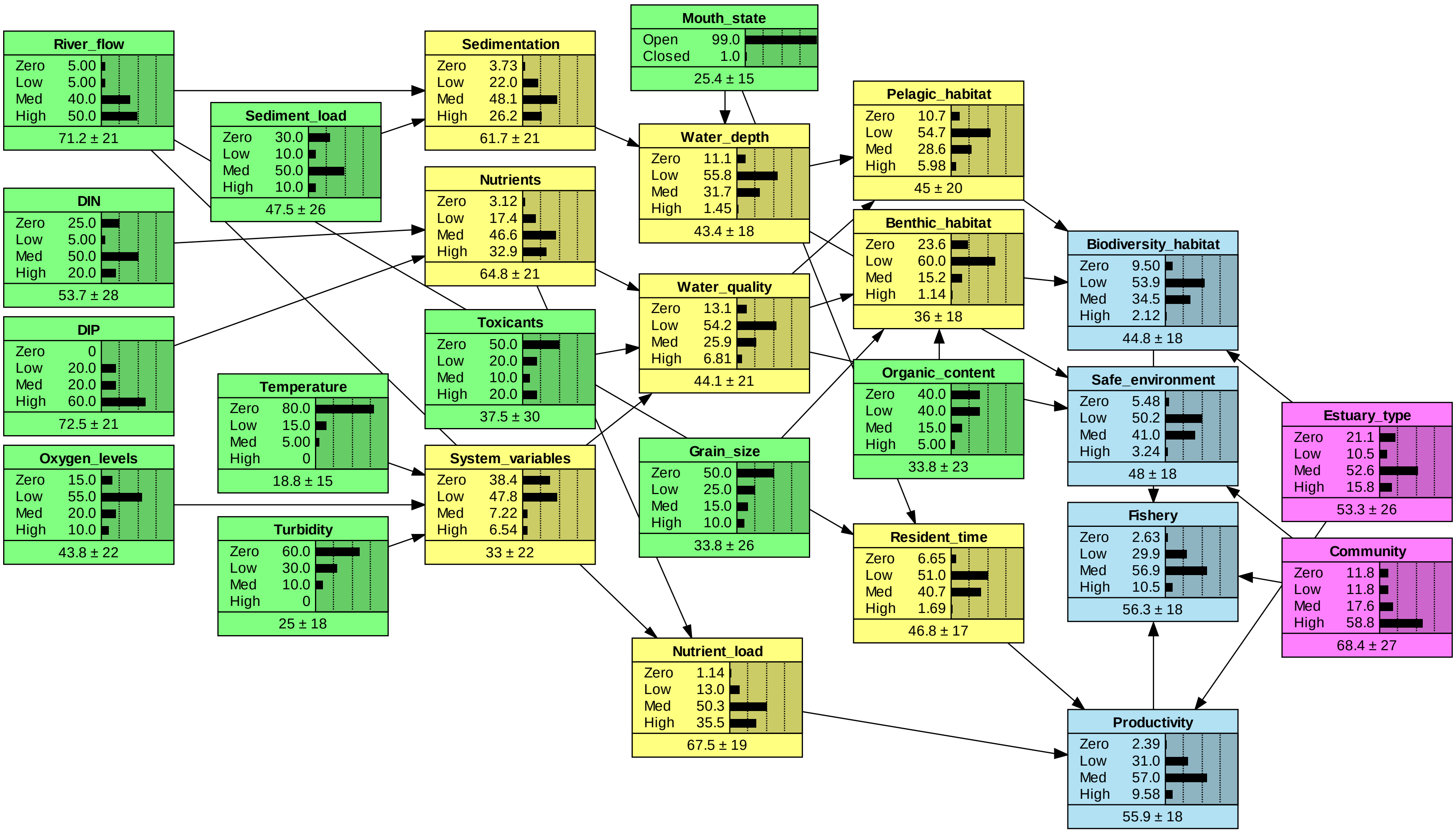

Fig. S26: Bayesian network relative risk model for Scenario 5 in uMvoti Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

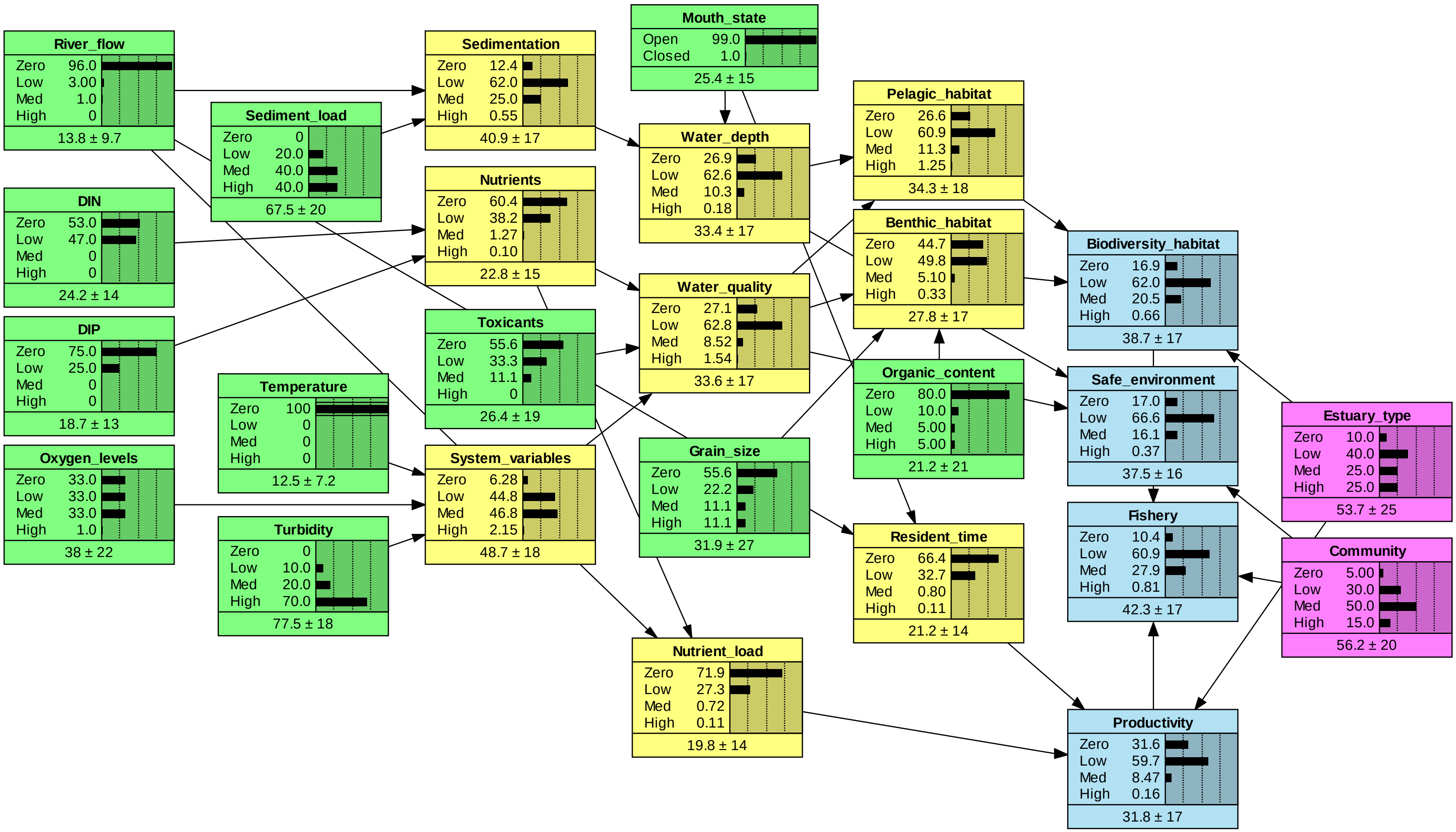

Fig. S27: Bayesian network relative risk model for Scenario 5 in Thukela Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

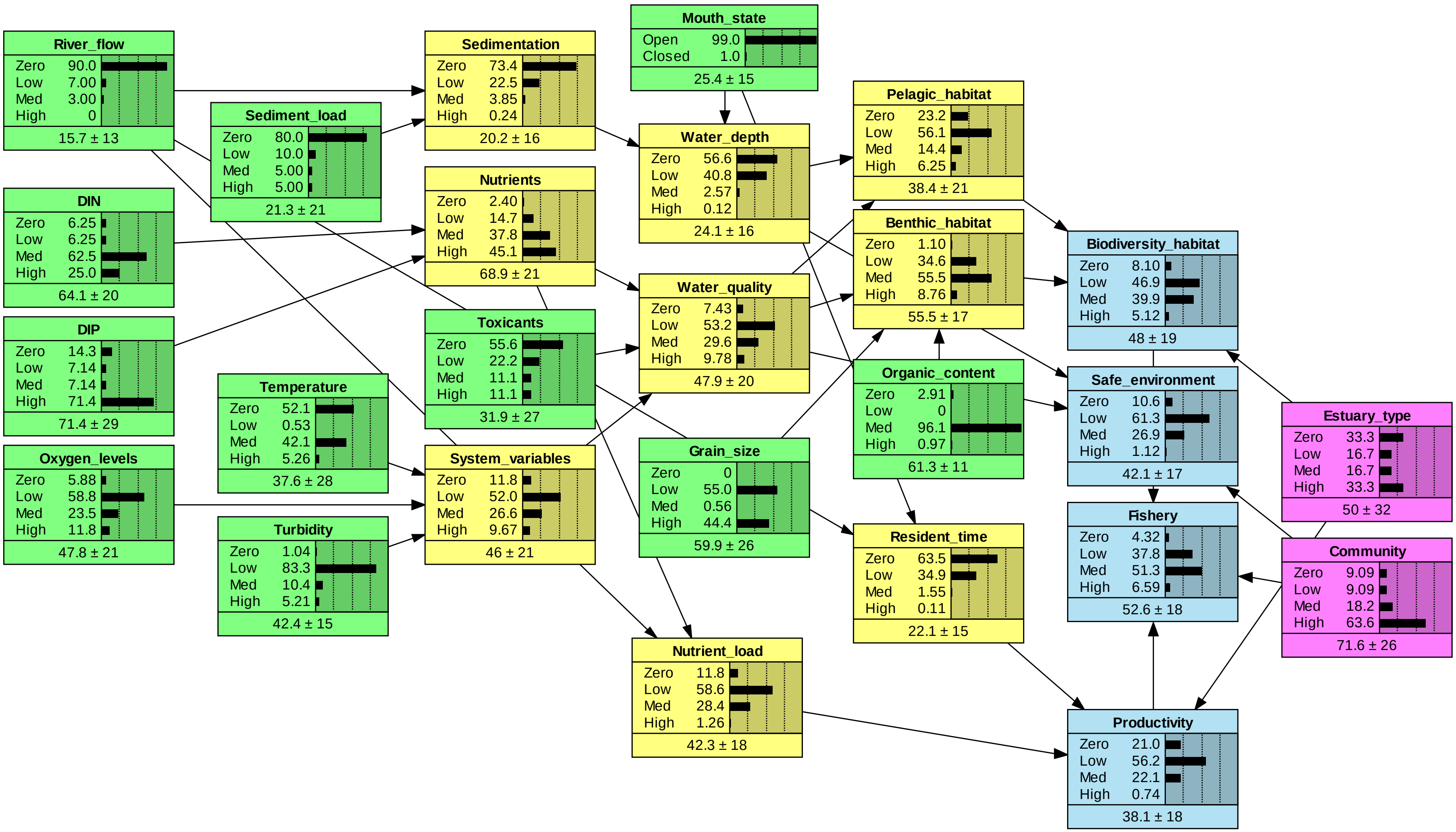

Fig. S28: Bayesian network relative risk model for Scenario 5 in Thukela Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

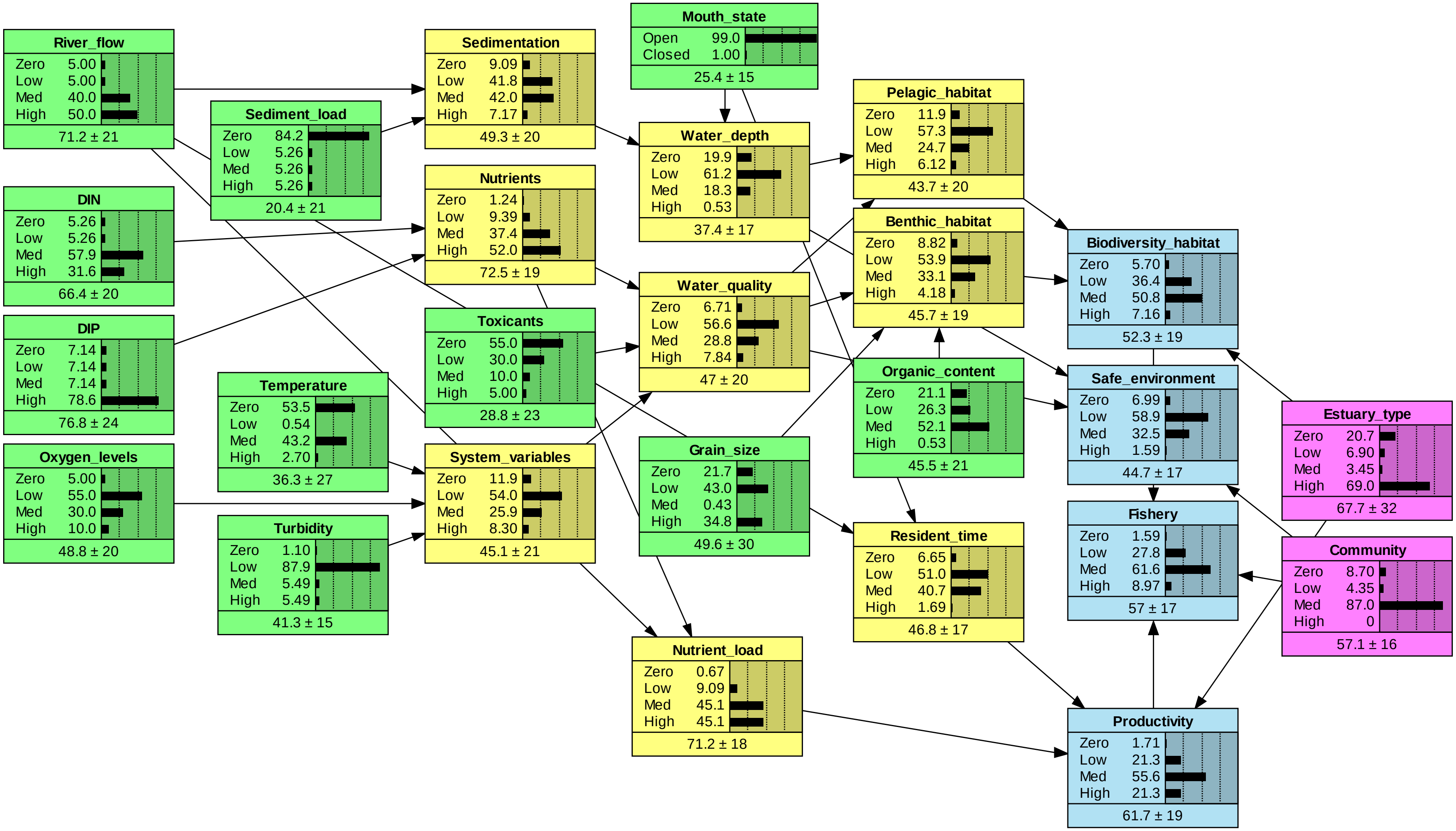

Fig. S29: Bayesian network relative risk model for Scenario 5 in aMatikulu/Nyoni Estuary (Low flow period). Bottom values in each node presents mean risk score ±SD.

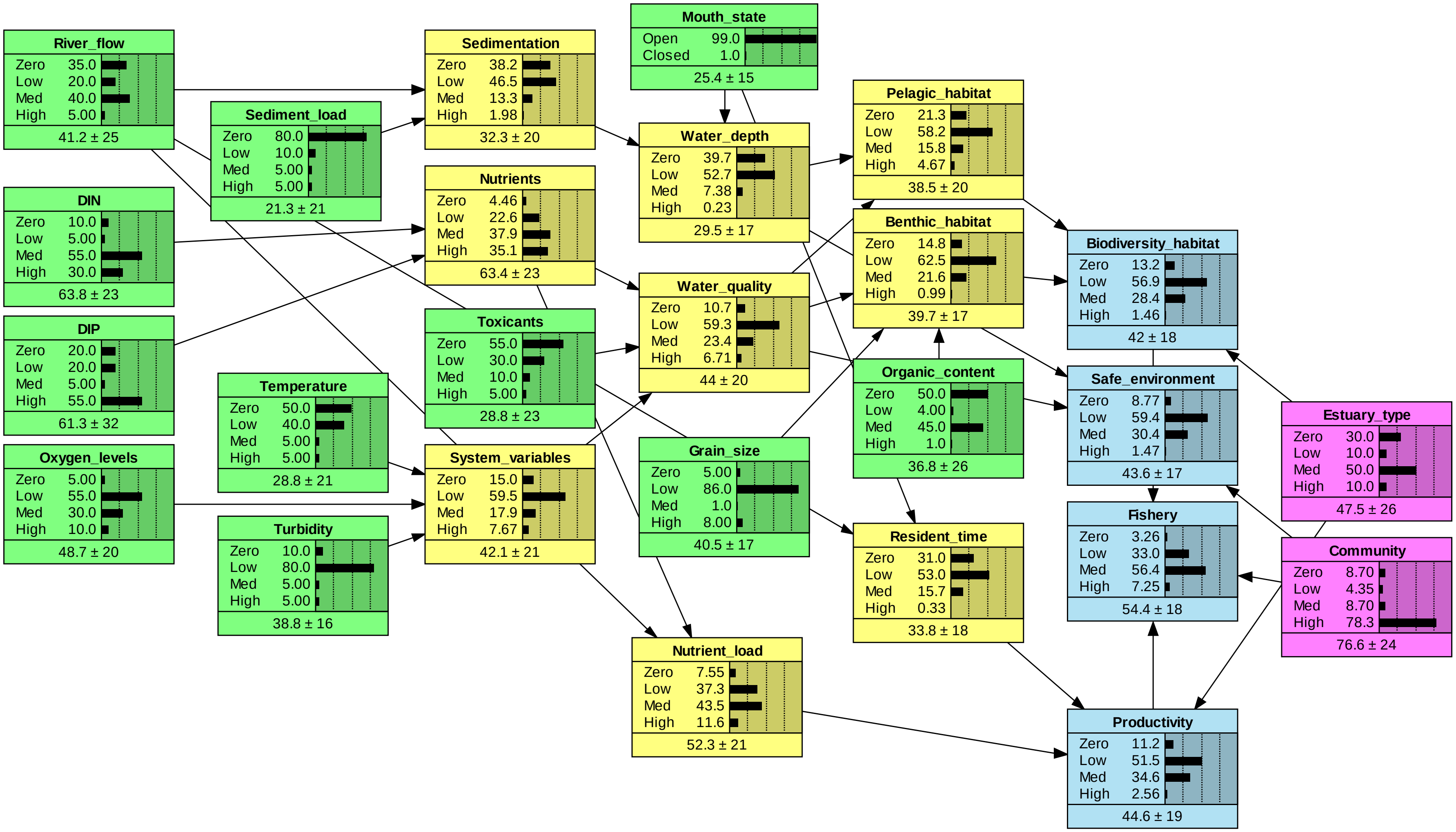

Fig. S30: Bayesian network relative risk model for Scenario 5 in aMatikulu/Nyoni Estuary (High flow period). Bottom values in each node presents mean risk score ±SD.

4.9 Supplementary material (S3)

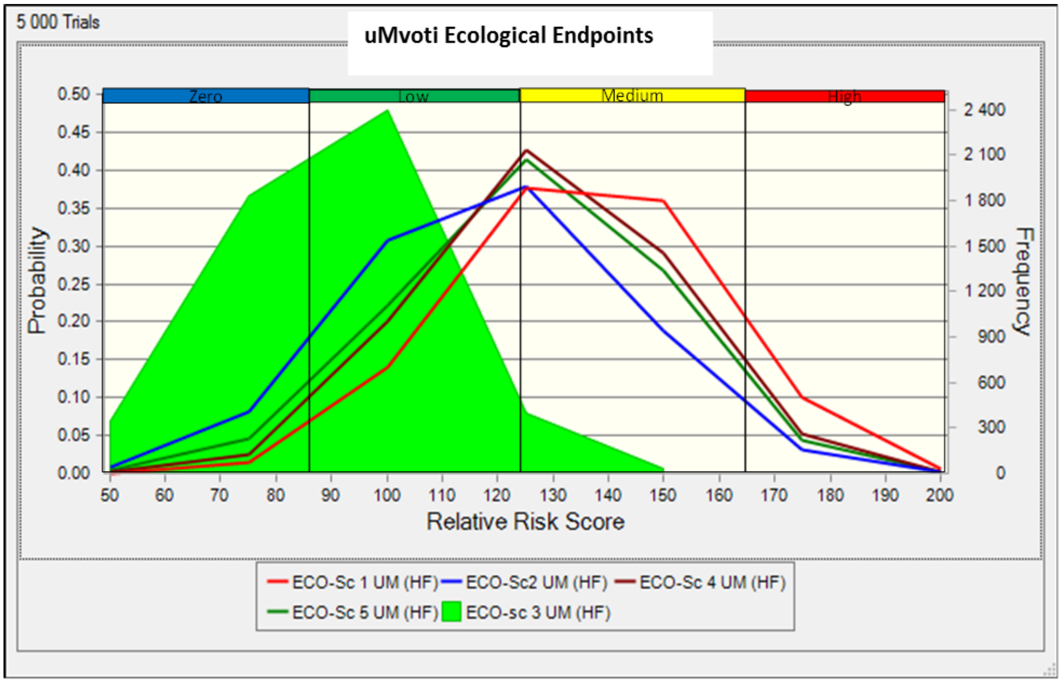

Fig. S1a: Uncertainty results for uMvoti ecological endpoints during high flow

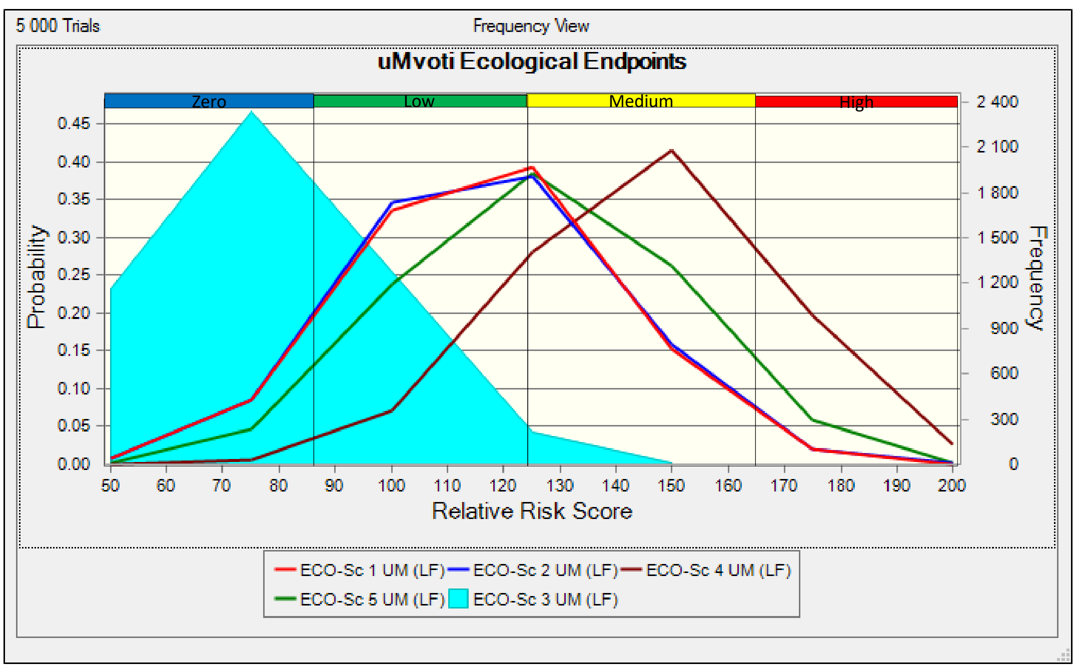

Fig. S2a: Uncertainty results for uMvoti ecological endpoints during low flow

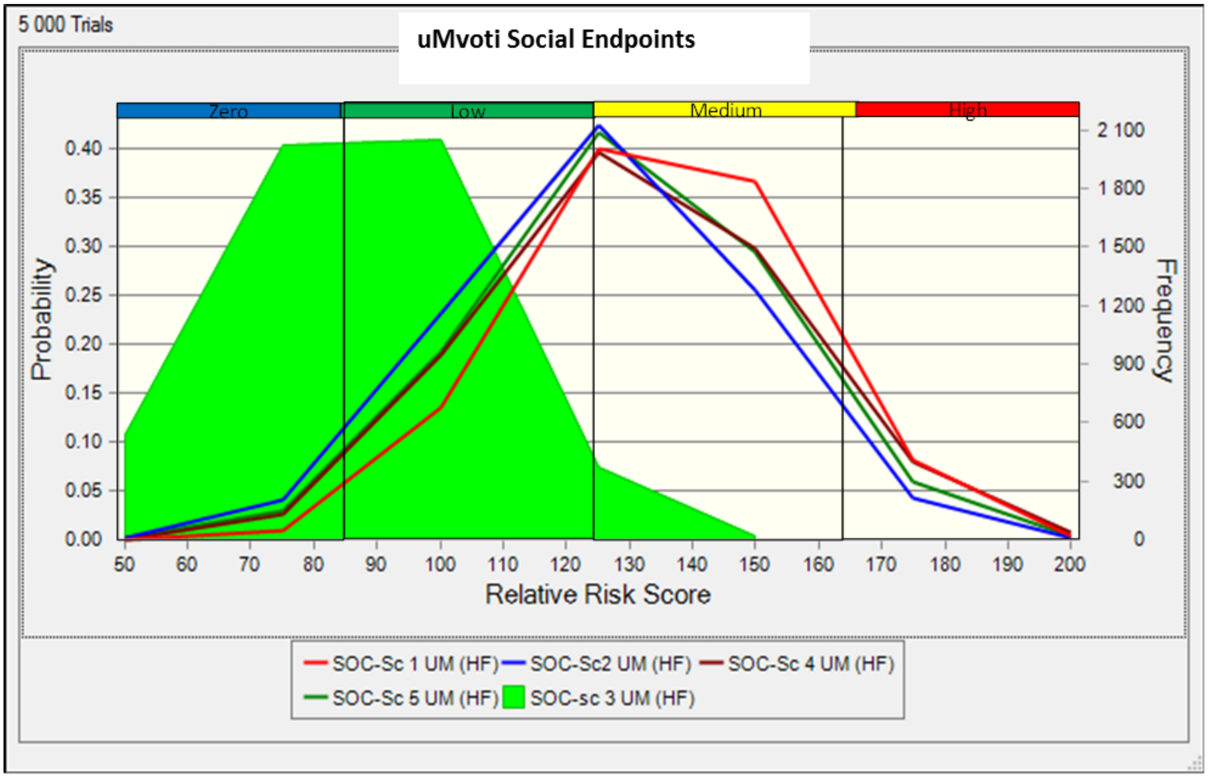

Fig. S3a: Uncertainty results for uMvoti social endpoints during high flow

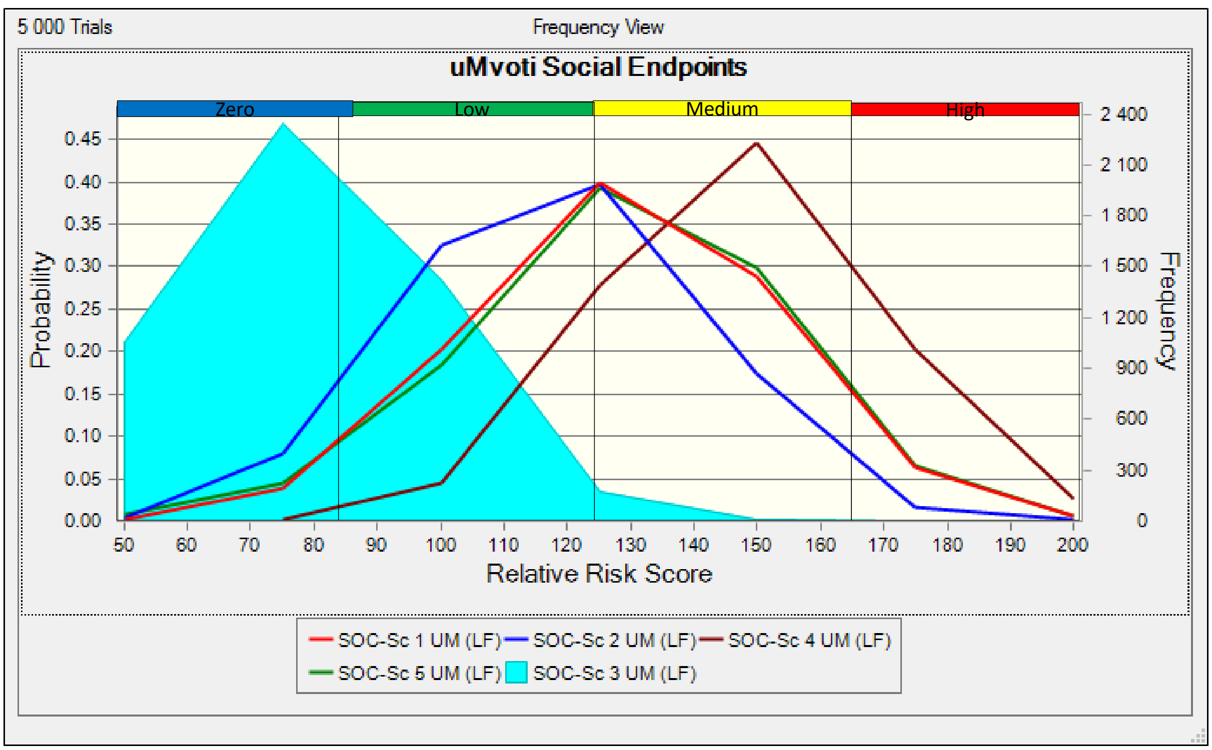

Fig. S4a: Uncertainty results for uMvoti social endpoints during low flow

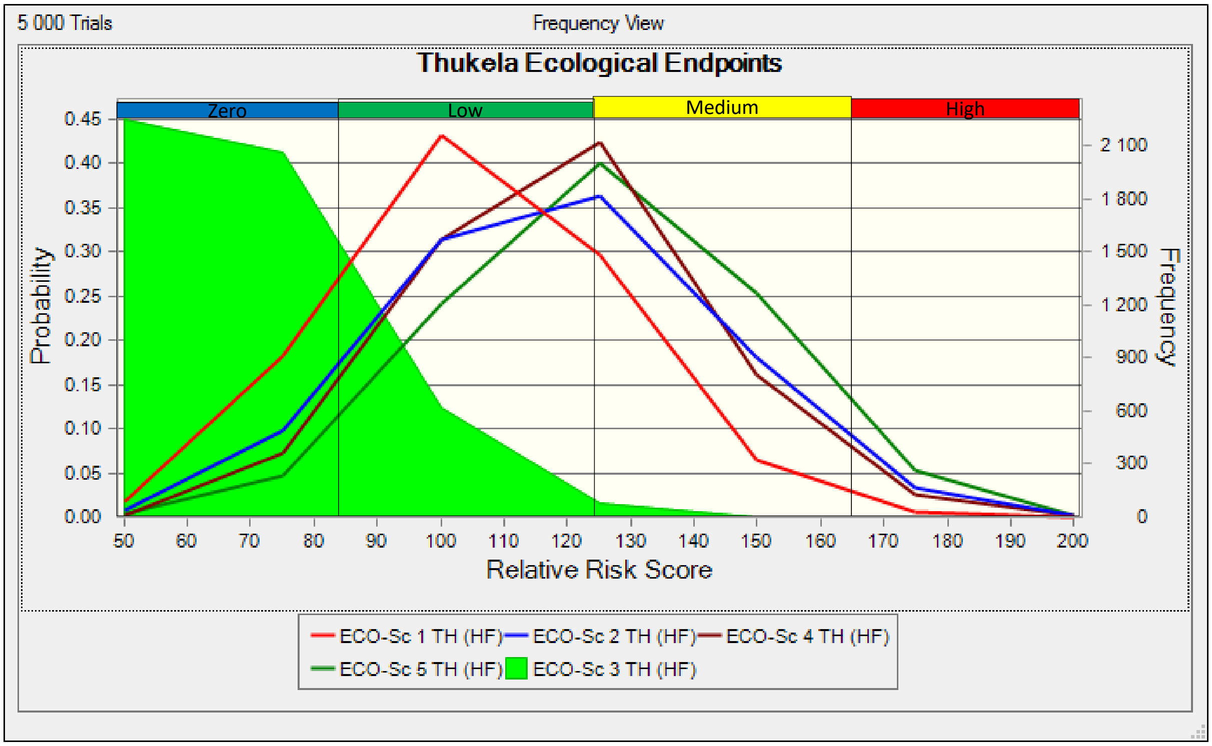

Fig. S5a: Uncertainty results for Thukela ecological endpoints during high flow

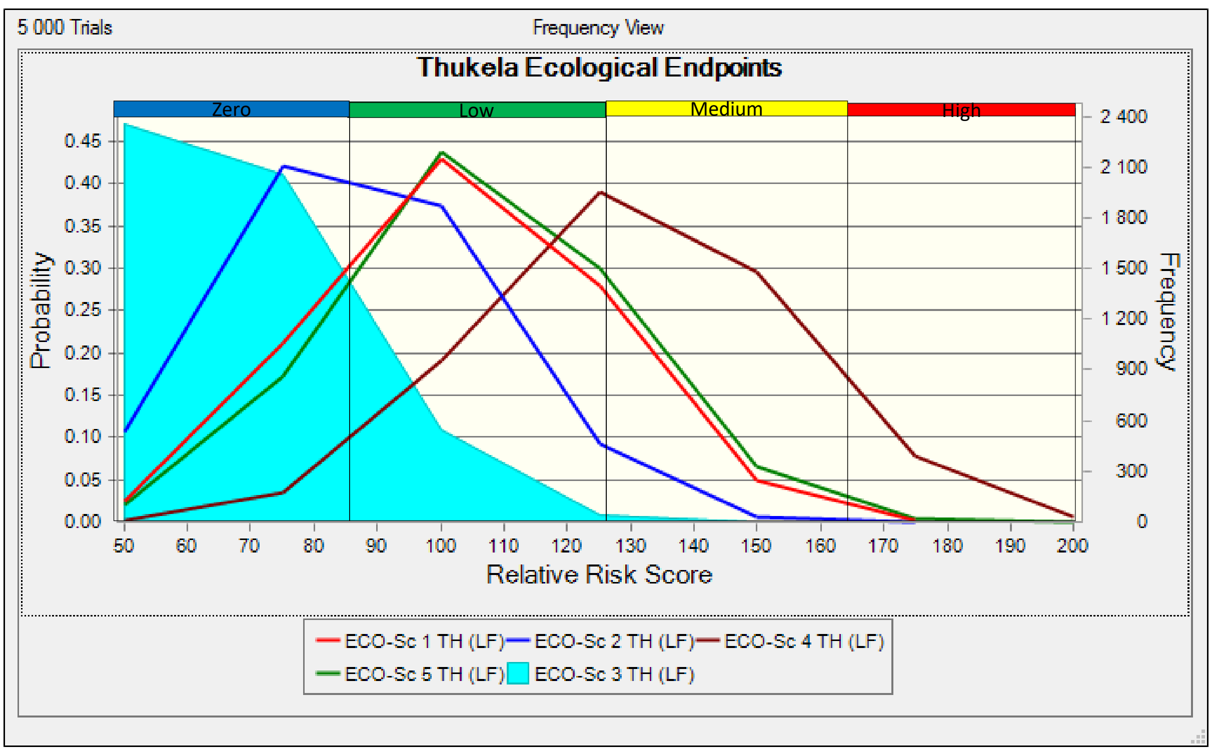

Fig. S6a: Uncertainty results for Thukela ecological endpoints during low flow

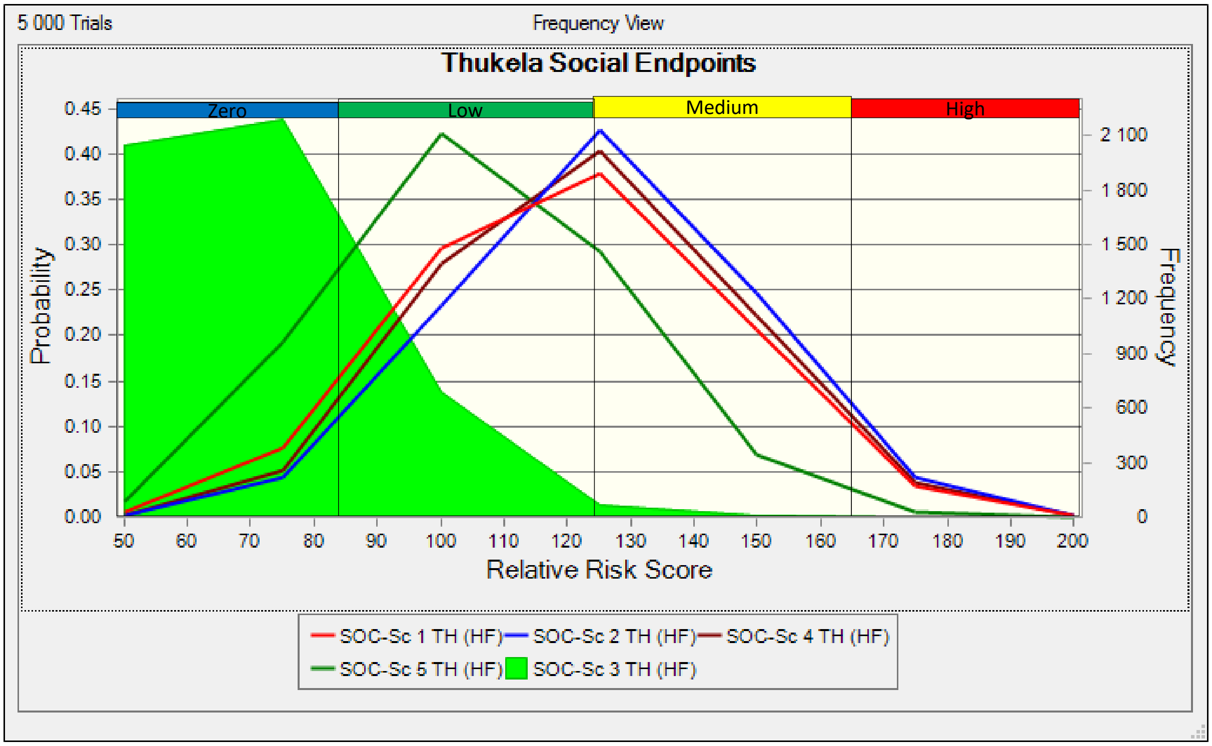

Fig. S7a: Uncertainty results for Thukela social endpoints during high flow

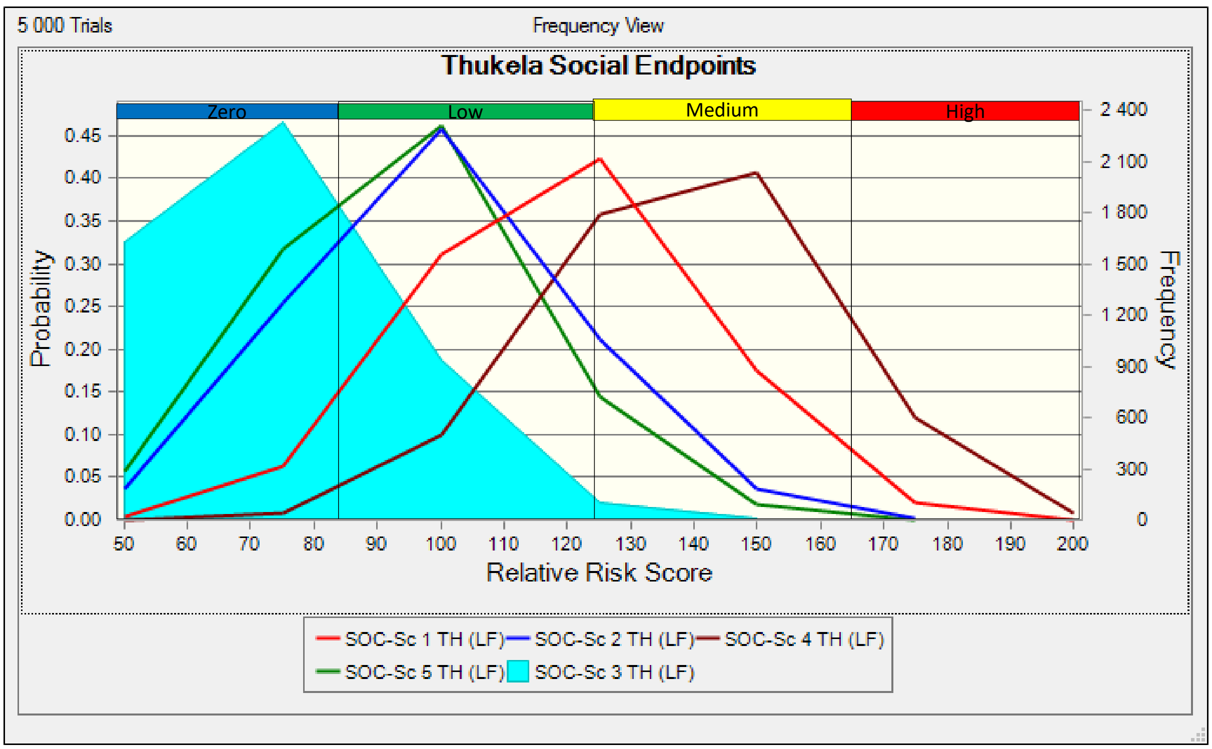

Fig. S8a: Uncertainty results for Thukela social endpoints during low flow

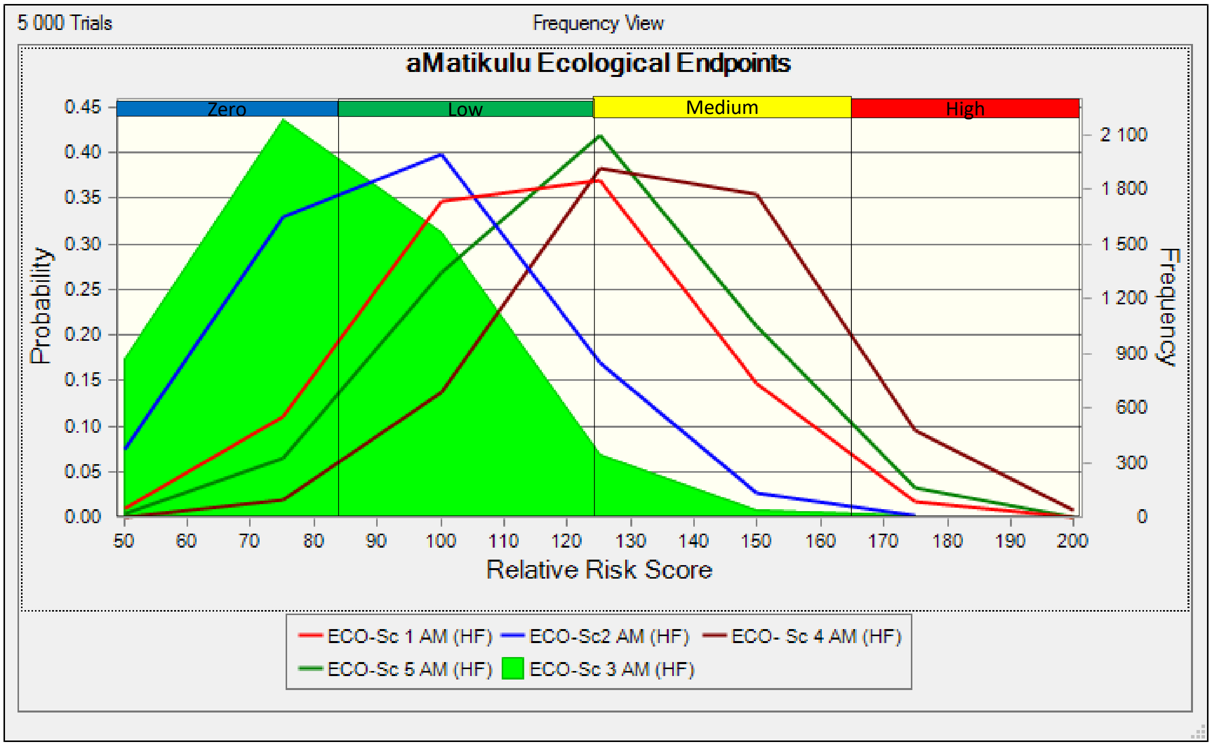

Fig. S9a: Uncertainty results for aMatikulu ecological endpoints during high flow

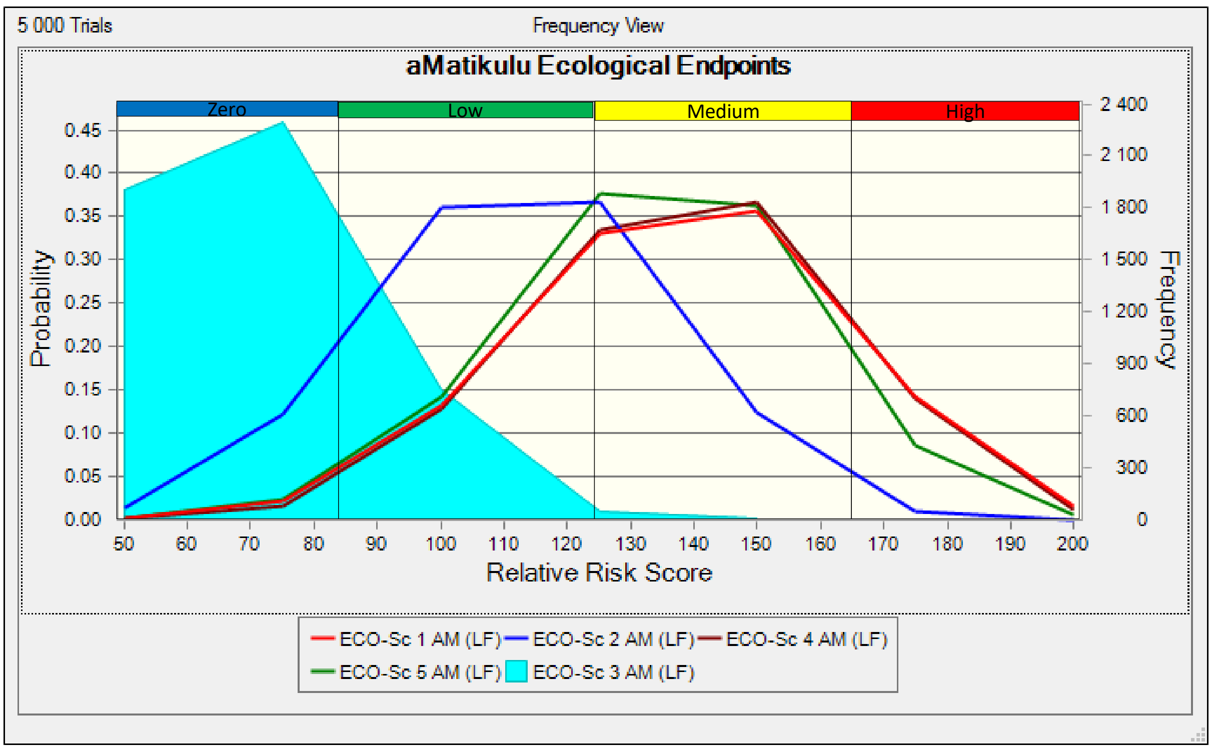

Fig. S10a: Uncertainty results for aMatikulu ecological endpoints during low flow

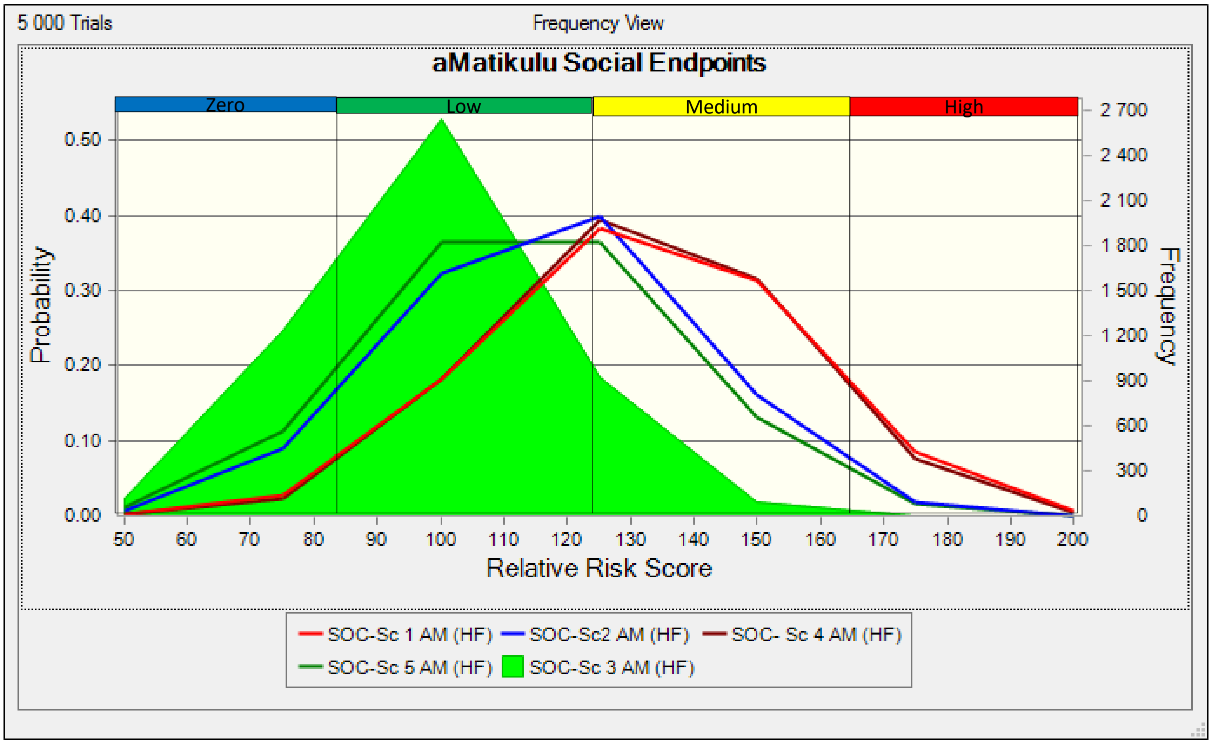

Fig. S11a: Uncertainty results for aMatikulu social endpoints during high flow

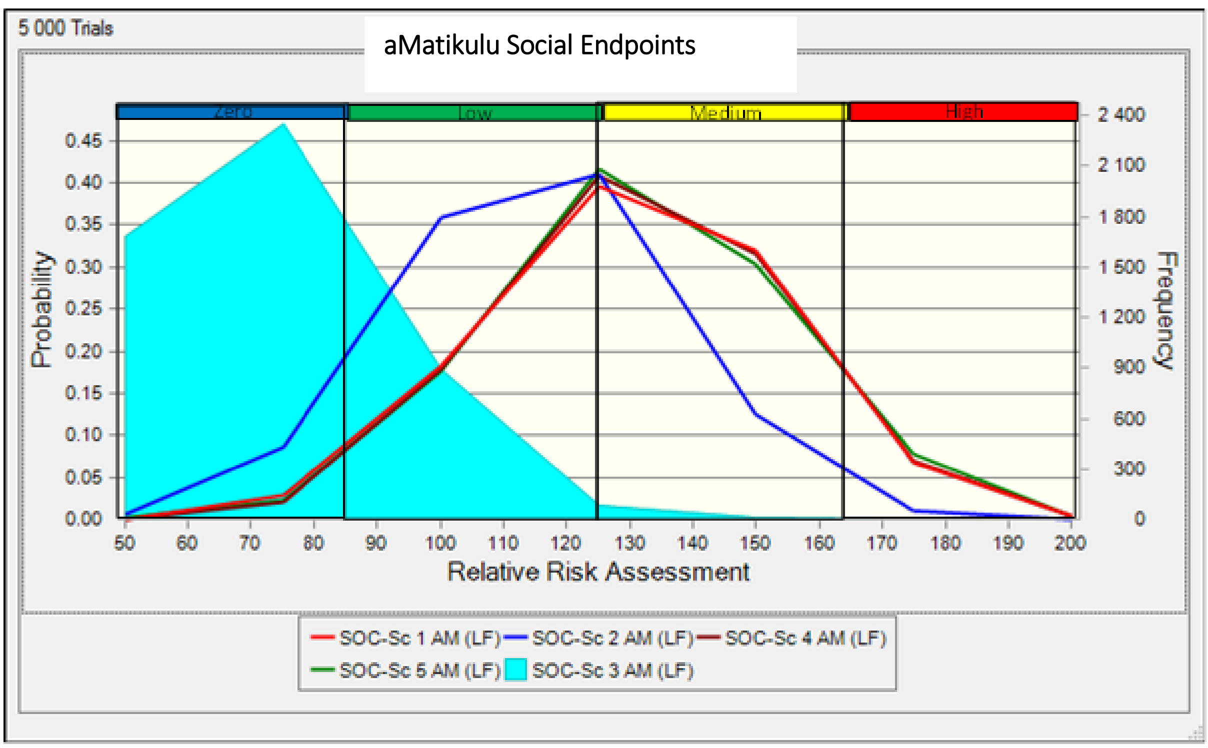

Fig. S12a: Uncertainty results for aMatikulu social endpoints during low flow

CHAPTER 5

Conclusion

5.1 Conclusions

This chapter summarises and discusses the main findings of the research in relation to the aim and objectives of the present study. Based on the outcomes of the study, management and conservation recommendations are also presented. In addition, restoration and rehabilitation suggestions for the catchments of uMvoti, Thukela and aMatikulu/Nyoni estuaries are also provided. The continued deterioration in the ecological state of South Africa's surface aquatic ecosystems, including these estuaries is causing an inevitable decline in the provision of key ecosystem services upon which the social and economic development of the country depends (Ashton, 2007; Driver et al., 2005). The information from the current study will contribute to the establishment of suitable balance between the use and protection of the estuarine ecosystems for municipal, district and national regulators.

In the current study, some physico-chemical parameters between the three estuaries were variable and identified as issues of particular concern. Although the three estuaries studied occur in the same geographical area and are geomorphologically similar according to Harrison et al. (2000), the variability in water quality conditions can be attributed to different land use activities and their varying intensities on these systems. Variability in invertebrates (macrozoobenthos and zooplankton) community structures in these three estuaries are also likely to be attributed to these alterations in water quality in particular and habitat conditions. Most environmental parameters were at more acceptable levels in the aMatikulu Estuary compared with the uMvoti and Thukela estuaries. Threats to the aMatikulu/Nyoni Estuary are generally minimal with little anthropogenic impact on the estuary currently which includes only one sugar mill and associated agricultural activities upstream of the aMatikulu River. The uMvoti Estuary, however, is largely affected by pollution from different anthropogenic and land use sources further upstream including effluent input of treated sewage, sugar and paper mill effluents from Gledhow Sugar Mill and Sappi Stanger Mill, agricultural irrigation and several domestic uses by urban and informal settlements (Tharme, 1996. MacKay et al., 2000; O'Brien et al., 2009; Swemmer, 2009)

The anthropogenic land use activities associated with the Thukela catchment include water abstraction for industrial and domestic use, industries, agriculture, mining, recreation, waste water treatment works, paper mill, and road and rail networks. Land use activities associated with the uMvoti and Thukela estuaries are likely to have resulted in the reduction of oxygen levels in these systems observed during the current study. These reductions can be attributed to major alterations in the ecological functioning of these systems. Large variation in turbidity levels between the three estuaries were also observed with the highest turbidity values recorded in the Thukela Estuary and the lowest recorded in the aMatikulu/Nyoni Estuary. This is evident in the Thukela River Estuary with high load of sediments accumulated in the lower reaches of the estuary. Such sediment accumulation is attributed to the poor flushing of the river as a result of reduced flow. This sediment accumulation thus increased the turbidity levels in the Thukela Estuary. The pressure on the structure and function of the Thukela system has recently increased due to elevated demand for water related ecosystem services from the Thukela River catchment (Pienaar, 2005). Although there is an increased pressure on the structure and function of the Thukela system as a result of increasing demand for water related ecosystem services, the lower portion of the Thukela River and its associated estuary are still characterized as an ecologically important region that provides a range of services.

The uMvoti Estuary catchment is heavily affected by the anthropogenic water resource use activities as described above. The impacts of these activities is reflected in the water quality and habitats of this system which in turn is reflected in the macrozoobenthos and zooplankton communities comprising low diversities and abundances. The uMvoti riverine system has been modified completely, with nearly total loss of natural habitat and biota and the destruction of many basic ecosystem functions (Tharme, 1996). As a result, the uMvoti Estuary is regarded as a degraded system which functions differently from the way it did in its former pristine state (MacKay et al., 2000). Since 1964, water quality of the uMvoti Estuary was described as grossly polluted (Begg, 1978). This may explain the dominance of the insects Chironomidae in the uMvoti Estuary. In addition, chironomid larvae are good indicators of pollutants as they are resistant to higher levels of water quality disturbance (Day, 1981; Hawking and Smith, 1997).

Low abundance of zooplankton during high flow in the uMvoti and Thukela estuaries could be a result of outflow of estuarine water washing away zooplankton into the adjacent sea. Zooplankton abundance was generally higher in the lower reaches of the uMvoti and Thukela estuaries during the current study. Higher zooplankton abundance in the lower reaches of the Thukela Estuary correlated with higher phytoplankton biomass (chl–a) on this region of the estuary. This relationship suggested that zooplankton abundance may be controlled by phytoplankton on this system. Most estuaries in South Africa experience minimum zooplankton abundances during low flow and maximum during high flow (Wooldridge, 1999) and zooplankton abundance in the aMatikulu/Nyoni Estuary during high flow was consistent with this pattern. Although the uMvoti had the higher chl–a concentrations when compared with the Thukela and aMatikulu/Nyoni estuaries, the lower zooplankton abundances are likely to have attributed to the insufficient residence time in this system. Higher zooplankton abundance in the aMatikulu/Nyoni Estuary was likely to have attributed to the relatively higher phytoplankton biomass, higher nutrient levels as well as sufficient residence time. Although the uMvoti and Thukela estuaries are affected by relatively similar land use activities and although these two systems are impacted by water abstraction activities up stream, the lower phytoplankton biomass (chl–a) in the Thukela Estuary is likely to have been a result of higher turbidity levels in this system. Highly turbid waters of the Thukela Estuary limit light penetration due to reduced clarity and this may prohibit primary productivity. Findings of the current study suggested that these systems require appropriate management to ensure adequate freshwater supply so as to maintain ecological structure and function of these ecosystems.

Ecological risk assessment has been widely known as an important tool in improving environmental decision making (Yu et al., 2015). Results from the RRM in the current study provided useful information that can be used by the environmental managers in establishing management measures and controlling/minimizing dominant stressors and thus protecting or restoring important habitats and endpoints in the three estuaries studied. Relative risk results showed that the endpoints in the three estuaries were exposed to the lowest risk during scenario three which was a scenario before major resource development. Results of the present study also revealed that if mitigation and management measures are implemented, risk probabilities will be low in the near future (i.e. scenario 5) for all the endpoints in the three estuaries studied. Risk results suggested that there has been a change in the estuarine systems studied over time. Such change is likely to have been brought by the increase in human population and industrialization and agricultural plantations in the catchments of the estuaries studied. Overall risk results across all scenarios showed that all endpoints were at a highest risk in the uMvoti followed by aMatikulu/Nyoni Estuary while the Thukela displayed lower risk to all endpoints. Although the aMatikulu/Nyoni Estuary is in a relatively good ecological state, results of the current study showed that this system requires management focus just as uMvoti Estuary. It was suggested that if the endpoints in this system are exposed to high risk, the sensitivity of this system will increase making it more vulnerable to external stressors. Results of the current study showed that some endpoints are at higher risk during low flow when compared with high flow with fishery and biodiversity habitat being two examples. Such results further highlighted the importance of the reduction in flows of these estuaries that will further pose more risk to the endpoints. This suggested higher system vulnerability and sensitivity when flows are reduced, and further raised the need to look at the flow requirements of these systems and implementation of laws in order to maintain acceptable flows and maintain ecological integrity of these estuarine systems. One of limitations in implementation of finding of RRM is the great uncertainty resulting from lack of data and poor quality of data (Yu et al., 2015). Additional data are thus required in the uMvoti, Thukela and aMatikulu/Nyoni estuaries to reduce uncertainty in the results. These data can be obtained through long term ecological monitoring of these systems so as to get the updates of the ecological states of these systems over time.

The anthropogenic land use activities that are associated with the three estuaries studied differed. Although geomorphology and biogeography of these estuaries are comparable, different land use practices with different intensities affected the ecological health of these systems differently. We showed that macroinvertebrates and zooplankton communities respond to stressors associated with land use activities. Alteration of water quality and quantity as a result of land use activities was evident in the present study. Habitat alteration was also evident in the current study. In the case of uMvoti Estuary, excessive use of water resources for anthropogenic activities is likely to have resulted in significant changes in the macroinvertebrate and zooplankton community. Disrupted ecosystem processes and potentially the functioning of this system might have resulted in these alterations in invertebrate communities. High loads of soft sediments in the Thukela Estuary, coupled with reduced flow is concerning and this condition may affect primary productivity in this system. Management intervention is urgently needed for these estuarine systems. Estuary Management Plans need to be developed and implemented for the protection of these estuarine systems in KZN with the Thukela Estuary being the priority followed by aMatikulu/Nyoni Estuary. The Thukela Estuary is an important system as it holds high ecological and economic value. With so many stressors acting on its catchment, the management measures in this system are needed to protect its diversity and ecological functioning. The aMatikulu lies within the nature reserve and displays higher species richness and abundances of macrozoobenthos and zooplankton. Increase in sediment load together with new changes in community structures suggested that new stressors are acting on its catchment. This estuary needs to be conserved and monitored to prevent stressors from affecting the community structures and functioning of this system. Results of the EcoRA showed that the aMatikulu/Nyoni is at high risk and this system is at a relatively pristine state when compared with the Thukela and uMvoti Estuary. Biological communities are more vulnerable to changes in land use activities that might change the abiotic environment. Generally, risk results suggest that this system is vulnerable to change. High impacts in the uMvoti Estuary have resulted in shifts in communities, loss of biodiversity, disturbance in habitat and nursery function. Although this system is heavily threatened and polluted, management measures are needed to improve water quality, biodiversity and regain nursery function. If no management and conservation measures are implemented in these systems, they will continue to deteriorate as a result of land use activities in their catchments and this might lead to a complete loss of nursery function as already seen in uMvoti Estuary, loss of biodiversity and severely altered water quality. Restoration of riparian vegetation of the estuaries studied can aid in improving water quality and aquatic habitats of these systems. Development of riparian buffers may be another important strategy to reduce sediment loading and erosion into these impacted estuaries.

5.2 Degree to which the objectives and hypotheses were met

We established three research objectives to understand how macrozoobenthos and zooplankton respond to altered water quality and quantity and to further identify the endpoints that are exposed to risk (and the extent of risk) in the uMvoti, Thukela and aMatikulu estuaries. Available information on macrozoobenthos and zooplankton in the uMvoti, Thukela and aMatikulu estuaries was sparse.

The first objective was to quantify and compare the composition (spatial and temporal trends) of macrozoobenthos communities within and between the uMvoti, Thukela and aMatikulu estuaries using multivariate statistical analyses. Macrozoobenthos communities are principal components in the functioning of estuarine ecosystems as they serve as food source for organisms in higher trophic level e.g. prawns and fish (Barry et al., 1996; Gray and Elliot, 2010;

Herman et al., 1999). These organisms are suitable ecological indicators in detecting the effects of stress and pollution (Patricio et al., 2009; Salas et al., 2006) as well as water and sediment quality (Chapman and Wang, 2000; Dauer and Ranasinghe, 2000). The Thukela Estuary had the highest number of taxa (n = 24) followed by the aMatikulu/Nyoni (n = 11) and then uMvoti Estuary (n = 8). However, the aMatikulu/Nyoni Estuary displayed the highest abundance (31 764 no·m^{-2}) when compared with other estuaries studied. Following aMatikulu/Nyoni Estuary, the Thukela Estuary displayed abundance of (29 589 no·m^{-2}) followed by the uMvoti Estuary (10 336 no·m^{-2}). Generally, the highest abundance was recorded during low flow in all estuaries studied. The abundance was higher in the upper reaches in both uMvoti and Thukela estuaries with no clear pattern in the aMatikulu/Nyoni Estuary. The RDA triplot revealed that higher temperature, turbidity and phosphate contributed to the structuring of the macrozoobenthos community in the uMvoti and Thukela estuaries while in the aMatikulu/Nyoni Estuary higher salinity, pH and oxygen were responsible for structuring the macrozoobenthos community (Chapter 2). The hypothesis that the macrozoobenthos community within and between the three estuaries would vary along the spatio-temporal scale, and that the differences would be related to variations in catchment land use patterns and water quality was accepted.

The second objective was to quantify and compare the composition (spatial and temporal trends) of zooplankton communities within and between the uMvoti, Thukela and aMatikulu estuaries using multivariate statistical analyses. Zooplankton are an important food source for many fish species and these organisms play a significant role in energy transfer from primary producers to secondary production (Harrison and Whitfield, 1990; Whitfield, 1998; Whitfield, 1985; Wooldridge and Bailey, 1982). The Thukela and aMatikulu/Nyoni estuaries had higher number of taxa (n = 10) when compared with the uMvoti Estuary (n = 5). The aMatikulu/Nyoni Estuary exhibited highest zooplankton abundance (15086.9 ind. m^{-3}) followed by Thukela (955.3 ind. m^{-3}) and then uMvoti Estuary (456.5 ind. m^{-3}). Zooplankton abundances in the uMvoti and Thukela estuaries were higher during low flow as opposed to the aMatikulu/Nyoni Estuary which exhibited higher abundance during high flow. The RDA plot revealed that higher salinity, conductivity and oxygen contributed to the structuring of the zooplankton community in the aMatikulu/Nyoni Estuary while turbidity and pH contributed to the structuring of zooplankton community in the uMvoti and Thukela estuaries (Chapter 3). The hypothesis that the zooplankton community within and between the three estuaries would vary along the spatio-temporal scale, and that the differences would be related to variations in catchment land use patterns and water quality was accepted.

The third objective was to carry out a Regional Scale Risk Assessment for the uMvoti, Thukela and aMatikulu estuaries to evaluate the threat of land use activities to selected protection endpoints for the study area. This study was the first to assess the risk of stressors to the socio-ecological endpoints of the uMvoti, Thukela and aMatikulu estuaries using the Bayesian network Relative Risk Model (RRM) approach. Endpoints considered in the current study were biodiversity habitat, safe environment, fishery and productivity. Results showed that the productivity generally displayed lower risk when compared with other endpoints in all estuaries, during all scenarios except for scenario 5, and during both low and high flows. Overall scenario 3 which is a scenario before major resource development in the study area had the lowest scores of risk for all the endpoints with the Thukela estuary displaying the lowest scores. The scenario that displayed highest risk scores to endpoints was scenario 4 which predicted risk scores to each endpoint for year 2025 if no laws and management measures are implemented with biodiversity habitat displaying the highest risk scores. The summation of all risk scores across all scenarios revealed that all selected endpoints were at a highest risk in the uMvoti Estuary followed by the aMatikulu/Nyoni Estuary and then the Thukela Estuary. Safe environment was most likely to be affected in the uMvoti Estuary while biodiversity habitat was most likely to be affected in the Thukela Estuary. In the aMatikulu Estuary, productivity and safe environment were most likely to be affected. In the uMvoti and Thukela estuaries, higher risk was associated with social endpoints indicating that people are at greater risk that the ecological components of these systems. However, in the aMatikulu/Nyoni Estuary, the ecological components of this systems were at a greater risk than the people as ecological endpoints displayed higher risk than the social endpoints. The hypothesis that the protection endpoints will display different risk scores between the three estuaries as a result of different anthropogenic land use activities posing different stress levels was accepted.

5.3 Recommendations

This thesis attempted to show the response of macrozoobenthos and zooplankton to altered water quality and quantity which resulted from land use activities in the catchments of the uMvoti, Thukela and aMatikulu estuaries. It also attempted to explain the risk distribution as a result of stressors from different sources to the selected endpoints in these systems and also the extent of risk to the individual estuarine systems. The results presented here have shown that the uMvoti Estuary is heavily impacted by the land use activities taking place in the upstream catchment. The impacts are reflected in the water quality of this system as well as biological communities. Such alterations were further witnessed in the risk analysis of this study where all the endpoints displayed the highest risk in this system when compared with other estuaries studied. The Thukela Estuary is also impacted by land use activities taking place upstream and these impacts are reflected in water quality and also the biological communities. In contrast, the aMatikulu Estuary still remained in a relatively good ecological state when compared with the uMvoti and Thukela estuaries in terms of abundances and species richness of biological communities although this system is likely to be at risk and thus need urgent management and protection. Development of Estuary Management Plans for these systems is essential so as to gather information and also to implement management measures. Biomonitoring alone is not enough to give insights of what endpoints are likely to be exposed to risk. Therefore, the environmental monitoring of these systems should be accompanied by EcoRA in which data essential to implement EcoRA are provided by environmental monitoring procedures. The EcoRA framework usually comes with uncertainty and this was also confronted in this research. Although these uncertainties depicted some limitations of the regional risk assessment framework and its application to the three estuaries studied, they did not affect the capacity of the RRM to give systematic means to evaluate and quantify the impacts of stressors to the endpoints on a relative basis which was the main goal of this study. To reduce uncertainty in the future EcoRA studies in these three estuaries, the following future works are proposed:

1. Future relative risk model studies in the uMvoti, Thukela and aMatikulu/Nyoni systems should focus on individual stressors and their contributions to the overall risk in each estuary or risk region.
2. Evaluation of specific sources for specific stressors is also necessary for future research to better understand and address the specific stressors in line with their specific sources.
3. Bayesian network RRM models established during the present study serve as a foundation for assessing the impacts of adaptive management strategies. Therefore, the upcoming regional risk framework studies in the three estuaries of the current study

should incorporate the development of common data platforms for these estuaries, refinement of the RRM and identification of data gaps.

4. Remediation and mitigation focus should be directed to multiple stressors and this will have higher efficiency on reducing risk to endpoints of the uMvoti, Thukela and aMatikulu/Nyoni estuaries. This means that when mitigation measures are implemented, they should not focus on one stressor of interest/concern, but multiple stressors should be considered as the RRM focuses largely on multiple sources, multiple stressors, multiple responses and multiple endpoints. This also highlights the necessity to consider all stressors to the endpoints not just those which initiated management action.
5. Future research should look at the response of these estuaries to the implementation of mitigation/ management measures. The preliminary RRM that was established during the current study will serve as a foundation for the future studies that will update our models. The results of the current study did show that if management actions are being taken for these estuaries, risk scores to all the systems and all endpoints will be reduced, however RRM thus needs to be updated once new data are available so as to validate the hypothesis.

5.4 References

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