Identification and prioritization of areas for riparian buffer restoration in the Sout River catchment, Overberg District, Western Cape

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Declaration

I declare that this thesis entitled '*Identification and prioritization of areas for riparian buffer restoration in the Sout River catchment, Overberg District, Western Cape*,' is my own work, and has not been submitted to any other university for any degree. All the sources I have used have been acknowledged by complete references.

Tshireletjo Manchidi

Full Name

Signature

09/03/2023

Date

Dedication

This work is dedicated to the Bogopa, Manchidi and Madiseng families for their unwavering support in my studies. I appreciate you all so much.

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Abstract

River catchments in areas dominated by agricultural activities are prone to contamination by nutrient-rich sediments whereby phosphorus is most concern. Phosphorus binds to fine sediments and gets transported to the rivers through surface runoff. Elevated levels of phosphorus in watercourse can compromise their water quality. One of the major consequences of high concentrations of phosphorus in watercourses is eutrophication which has detrimental effects on biota. Therefore, measures need to be put in place to protect water resources and prevent further water quality degradation.

Riparian vegetation buffers are essential for providing nutrient absorption functions that can limit the amount of pollution entering streams. They slow down sediment-laden runoff and may deposit or absorb sediments together with nutrients and pollutants attached to them. It is for this reason that they have been highly advocated for as one of the best management practices in reducing sediment contamination in watercourses. However, their effectiveness depends on their spatial placement within the catchment. Therefore, the aim of this study was to develop an approach that incorporates buffer effectiveness (sediment connectivity) and opportunity (adsorbed phosphorus concentration) to identify riparian areas that should be prioritized for buffer restoration in the Sout River catchment.

To achieve this aim, an integrated index of connectivity (IC) and vegetation cover (NDVI) model was computed to identify areas that have a high contamination risk. IC was assessed using the SedInConnect geospatial modelling tool. Additionally, Sediment samples were collected from slack water deposits and analysed for adsorbed total phosphorus concentration, particle size distribution and organic carbon percentage. This was done to determine the longitudinal variation in adsorbed phosphorus concentration in the catchment. The adsorbed phosphorus concentrations were then compared to background level concentration at a natural 'reference site'. Water samples were also collected and analysed for orthophosphate concentrations in Sout River.

The results show low IC values upstream and on the southern side of the catchment, whereas high IC values were observed on the north-eastern side and downstream of the catchment. Adsorbed phosphorus concentrations are lower upstream and higher downstream compared to background concentrations, suggesting that sediments get eroded upstream and get deposited downstream. No statistically significant correlations were found between adsorbed phosphorus

concentration and orthophosphate, organic matter, and particle size. This suggests that adsorbed phosphorus concentration in this catchment is influenced by factors other than these. Finally, the sediment-associated phosphorus contamination risk map shows that areas with high connectivity are associated with high concentrations of adsorbed phosphorus and vice versa. Therefore, these results suggest that areas that should be prioritized for riparian buffer restoration are those located downstream and on the north-eastern side of the catchment.

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Chapter 1: Introduction

1.1Background to the study

The water provisioning services offered by rivers and riparian zones often makes them prone to degradation because of human-induced disturbances such as water abstraction and damming as well as land use changes (du Plessis et al., 2022). Changes in river and land management practices have simplified the physical structure of riparian environments, altered river morphology, and degraded water quality over the last century, resulting in a decline in the quality and function of riparian ecosystems (Cole et al., 2020). It has become evident in recent years that these systems need better protection and management (Herdien et al., 2005).

Water quality is degraded when nutrients, toxicants, and pathogens are introduced into an aquatic system through both point source and non-point source activities (Zia et al., 2013). Point-source discharges originate from a single source and examples may be contaminants from sewage or industrial outlets. Non-point source fluxes, on the other hand, may come from diffuse sources with no discrete point of entry (Zia et al., 2013). Examples include cultivation and livestock farming, as well as peri-urban activity. The implications of poor water quality not only affect the aquatic ecosystems and associated biota, but also have impinged upon the continued sustainability of human populations and societies that rely on these resources (Blanche, 2002). As a result, water quality control agencies in various countries have implemented laws to reduce hazardous and chemical loadings from both point and non-point sources, however, non-point pollution remains a major issue that requires more research attention (Van der Laan and Franke, 2019).

Several legislations have been introduced to protect aquatic resources and prevent further degradation in South Africa. These include the National Water Act (NWA) 36 of 1998 (Department of Water Affairs and Forestry, 1998). The NWA is regarded as one of the most cutting-edge and comprehensive pieces of water management and control legislation in the world (Blanche, 2002). However, despite the existence of these policies, they still fail to adequately protect South Africa's aquatic systems and as a result, there is still a growing water scarcity and a decline in aquatic biodiversity (Macfarlane et al., 2009). This highlights the need to introduce additional approaches to ensure the protection of these aquatic systems.

Agriculture is one of the major non-point source pollutants which contributes to the degradation of water quality in watercourses. Common practices that cause this are bank erosion and sediment runoff which results from a reduction in natural vegetation, urine contamination, faeces, agrochemicals and veterinary antibiotics (Cole et al., 2020). Additionally, discharges are untreated, usually contain high loads of nutrient and organic matter, and they occur sporadically during intense rainfall events from sources that are not easily identifiable, quantifiable and controllable (Heathwaite and Dils, 2000). Unfortunately, due to the increasing population the demand for agricultural supplies has also increased and thus causing the industry to expand rapidly. Crop production has more than doubled in the past 50 years despite agricultural land cover only increasing by 10-15% (Schipanski and Bennett, 2012). This is due to increased fertilizer inputs, increased dependency on irrigation and the development of higher yielding crops (Schipanski and Bennett, 2012). Therefore, it is the management of nutrients and modern farming practices that has resulted in the current concern over the effects of agriculture on water quality.

Phosphorus (P) and Nitrogen (N) are the nutrients of main concern. They are the main nutrients that are used in fertilizers to increase crop yield and are essential for plant growth. Agriculture-driven increases in the mobilization of these nutrients have detrimental effects on the ecosystem. Their emission from agricultural lands is a major cause of a decline in water quality, dissolved oxygen concentrations and biotic population structure in rivers across the world (Bowes et al., 2005). Since the industrial revolution, anthropogenic activities have multiplied the mobilization of N and P into ecosystems globally (Allafta et al., 2020). Elevated levels of these nutrients eventually lead to eutrophication in water bodies where phosphorus was found to be the limiting nutrient for algal growth (Anderson et al., 2013). As such, successful attempts to limiting eutrophication in freshwaters typically involve decreasing P inputs (Kim et al., 2002; Tekile et al., 2015).

Eutrophication was found to be a major threat in many South African catchments (Van Ginkel, 2011). Villiers and Thiart (2007) studied the nutrient status in major South African river catchments using data from 1970 to 2005 and observed an alarming and statistically significant increases in dissolved phosphate concentrations in about 60% of these catchments. Eutrophication causes an increase in algal blooms, water turbidity, oxygen depletion, and the dominance of some species and eventually, a loss of biodiversity (Allafta et al., 2020). Moreover, oxygen depletion can result in the mobilization of heavy metals that were bound to sediments (Allafta et al., 2020). These metals have serious detrimental impacts on ecosystem health and water quality (e.g., Jonsson et al., 2003; Ip et al., 2007). Additionally, oxygen depletion in watercourses results in the death of aerobic plants and microbes (Singh, 2013).

Sediment is the main (85%) source of phosphorus enrichment in watercourses and the main mechanism by which phosphorus is delivered to surface water bodies (Jontos, 2004; Van der Laan and Franke, 2019). This is because of phosphorus' strong adsorption capacity to soil particles and organic matter (Van der Laan and Franke, 2019). Therefore, the characteristics of sediments in the catchment need to be studied to understand pollution problems linked with phosphorus in a catchment. The main pathways for phosphorus losses from agricultural lands were found to be surface runoff and erosion (Andersson et al., 2013). Excessive precipitation events result in high volumes of runoff which have the capacity to erode substantial amounts of sediments from hillslopes and deposit them near watercourses. However, not all eroded sediment will eventually reach the drainage basin. Sediments that are likely to reach the drainage basin are those that are eroded from sources that are well coupled to the drainage network and therefore sediment management strategies need to be targeted in these areas. Hence, using sediment as an environmental monitoring tool can provide an understanding of the sources and distribution of pollution sources throughout the catchment (Fredrick, 2001).

The pollution of watercourses caused by phosphorus used in fertilizers is a growing global issue. With the Sout River catchment being almost entirely covered by agricultural activities, there is an urgent need for action to control the concentrations and fluxes of this nutrient in the river. Technical and biological measures should be put in place as they increase efficiency and reduce costs (Zalewski, 2014). Vought et al. (1995) emphasized that existing knowledge in the development of management strategies should be utilized in conjunction with a change in agricultural practices to control the nutrient concentrations and fluxes in freshwaters. The re-establishment of riparian buffers is one of the strategies that is widely advocated for to control this pollution (Stutter et al., 2012; Poole et al., 2013). In agricultural landscapes riparian field margins occur in the transition zone between agriculture and watercourses, hence it is commonly recommended that riparian buffers be developed in these zones to control diffuse pollution caused by nutrients (Cole et al., 2020).

Buffers keep sediments from entering the channel network by disrupting lateral and longitudinal linkages within a catchment (Fryirs et al., 2007). Riparian buffers function as filters and sinks for sediments that would otherwise flow or be eroded into watercourses and degrade their water quality. They slow down sediment-laden runoff and may potentially deposit or absorb almost all the sediments together with nutrients and pollutants attached to them (Hawes and Smith, 2005). For this reason, riparian buffers have been identified as one of the best management tools in reducing sediment contamination in watercourses (Clausen et al., 2000; Abu-Zreig, 2004; Hickey and Doran, 2004; Dosskey et al., 2010). However, for the buffer to be effective there are several factors that need to be considered.

Catchment characteristics and the intrinsic characteristics of buffers have been identified as factors that influence buffer effectiveness. Of these, significant research attention has been given to intrinsic buffer characteristics such as buffer width and vegetation type (Dillaha, 1989; Hawes and Smith, 2005). There has been a considerable amount of research dedicated to determining design recommendations for vegetated buffer strips (e.g., Fischer and Fischenich, 2000). However, the process of determining the spatial placement of riparian buffers for water quality protection in a catchment is not well documented.

Identifying sites in the catchment where environmental benefits of buffers can be maximized is a crucial task, given the significance of sediment pathways (Tomer et al., 2003). Moreover, due to limited funding for restoration of public land, cost-effectiveness would be improved by targeting buffers to areas where they would be of the greatest benefit to the watercourse (Dosskey and Qui, 2010). For this reason, the current study proposes a method of identifying priority areas in the catchment for riparian buffer restoration. The proposed method incorporates buffer effectiveness and opportunity (Kotze et al., 2009). In this case, buffer effectiveness refers to the ability of the buffer to decouple catchment hillslopes from the drainage network by forming an impediment to sediment transfer, whereas 'opportunity' relates to the spatial distribution of adsorbed phosphorus concentrations in the catchment.

1.2 Research problem

The use of riparian buffers as a best management practice for diffuse pollution caused by sediment-laden runoff in watercourses is well studied and highly advocated for. However, there are limited studies that focus on developing a system of prioritization of riparian areas in the catchment for restoration, especially in developing countries such as South Africa with limited resources. This is a problem because buffer effectiveness is highly dependent on its location and should therefore be included in the planning of riparian buffer restoration.

1.3 Research question

Where in the catchment should riparian buffers be restored to achieve optimum buffer effectiveness?

1.4 Central argument

Optimum buffer efficiency can only be achieved if riparian buffers are restored in areas that are degraded and are strongly coupled to the drainage network, and where adsorbed P concentrations are high.

1.5 Study aim and objectives

Aim: To identify areas of the Sout River catchment that should be prioritized for riparian buffer restoration.

Objectives:

1. Determine the spatial variation in sediment connectivity in the Sout River catchment.
2. Determine the longitudinal variation of adsorbed Phosphorus concentrations in the Sout River.
3. Compare the concentrations of adsorbed Phosphorus with background levels at a natural reference site.
4. Identify high risk areas of adsorbed Phosphorus contamination in the Sout River catchment.

1.5 Rationale (Significance of the study)

Nutrient-laden sediments are increasingly degrading the water quality of rivers in agricultural dominated landscapes where fertilizers are applied in excess to meet the increasing demand for crop production. Phosphorus, which is one of the main nutrients of concern, builds up in soils due to its strong adsorption and immobilization into unavailable forms (Menezes-Blackburn et al., 2018), gets eroded by surface runoff and is deposited into watercourses degrading their water quality. This study emphasizes the use of riparian buffers as a best management practice for protecting watercourses from diffuse pollution caused by nutrient-laden sediments and proposes an approach that incorporates both buffer effectiveness (connectivity) and opportunity (phosphorus concentration) when prioritizing areas for riparian buffer restoration. The results from this study may be used in decision making and management of diffuse pollution caused by sediments in the study area, and the proposed approach can be applied in other areas as well, especially where there are limited resources and limited accessibility.

The Sout River flows through agricultural land along its entire length, thus making it susceptible to nutrient contamination. Nutrient contamination causes a decline in water quality which threatens aquatic biodiversity. Furthermore, the river drains into De Hoop Vlei which is a RAMSAR listed saline coastal lake. Contamination by nutrients from agricultural land has been recognized as a potential threat to the ecology of the lake (Lanz, 1997). For this reason, protection and management of the Sout River catchment is crucial.

The Overberg is home to the endemic Rûens Renosterveld vegetation which forms part of the Greater Cape Floristic Region (GCFR). The GCFR is popularly known as the smallest yet richest plant region on Earth. However, more than 90% of this vegetation type has been cleared to make way for agriculture (Curtis-Scott et al., 2020). As a result, Rûens Renosterveld is currently classified as Critically Endangered (Curtis-Scott et al., 2020). Restoring the riparian vegetation with the indigenous Rûens Renosterveld will contribute to its sustainability in the area.

Nutrient-laden sediments are increasingly contaminating water resources through diffuse pollution. It is difficult to control this pollution as it is diffuse (difficult to get where it comes from). As a result, there is a need to put measures in place to control them from being eroded into watercourses. The study identifies areas in the catchment that should be prioritized for riparian buffer restoration to control the erosion of sediments into the river channel. This approach can be adopted and used to control diffuse pollution caused by adsorbed phosphorus. The findings of this study can be used to inform management decisions when it comes to sediment management in the catchment.

1.6 Overview of the study

The motivation behind this study is to advocate for better water quality, especially in a developing country such as South Africa. Additionally, the study also advocates for the restoration of indigenous vegetation and protection of riparian areas. The overall purpose of this study is to propose a method that can be used to identify priority areas for riparian buffer restoration in a catchment. Riparian buffers reach optimum efficiency in reducing diffuse pollution if they are placed in areas that are highly connected to the sediment sources. Therefore, this study proposes an approach that combines buffer effectiveness (connectivity) and opportunity (adsorbed phosphorus concentrations) to identify such areas in the catchment.

The study aims at providing more knowledge and tools to encourage the use of riparian buffers in sediment management practices. Furthermore, the vegetation in the chosen study site has been listed as Critically Endangered and therefore needs urgent restoration and protection to ensure its sustainability. Thus, this study advocates for the restoration of this vegetation.

1.7 Research framework

Figure 1.1 below illustrates the framework of the present study. The study is aimed at identifying areas of the Sout River catchment that should be prioritized for riparian buffer restoration. To achieve the aim, the study focuses on four main objectives. The first objective focuses on determining the spatial variation in sediment connectivity in the catchment. The intension is to get an understanding of the areas in the catchment that are most likely to contribute the most sediments into the river channel. The second and third objectives focus on determining the longitudinal variation of adsorbed Phosphorus concentrations and comparing these concentrations to background levels at a 'reference' site. This is done to identify areas in the catchment that contain high concentrations of adsorbed phosphorus and therefore are likely to transfer it into the river. Finally, the fourth objective focuses on identifying areas in the catchment that should be prioritized for riparian buffer restoration based on the concentration of adsorbed phosphorus and sediment connectivity. Figure 1.1 provides a summary of the data collection and analysis methods used for each objective.

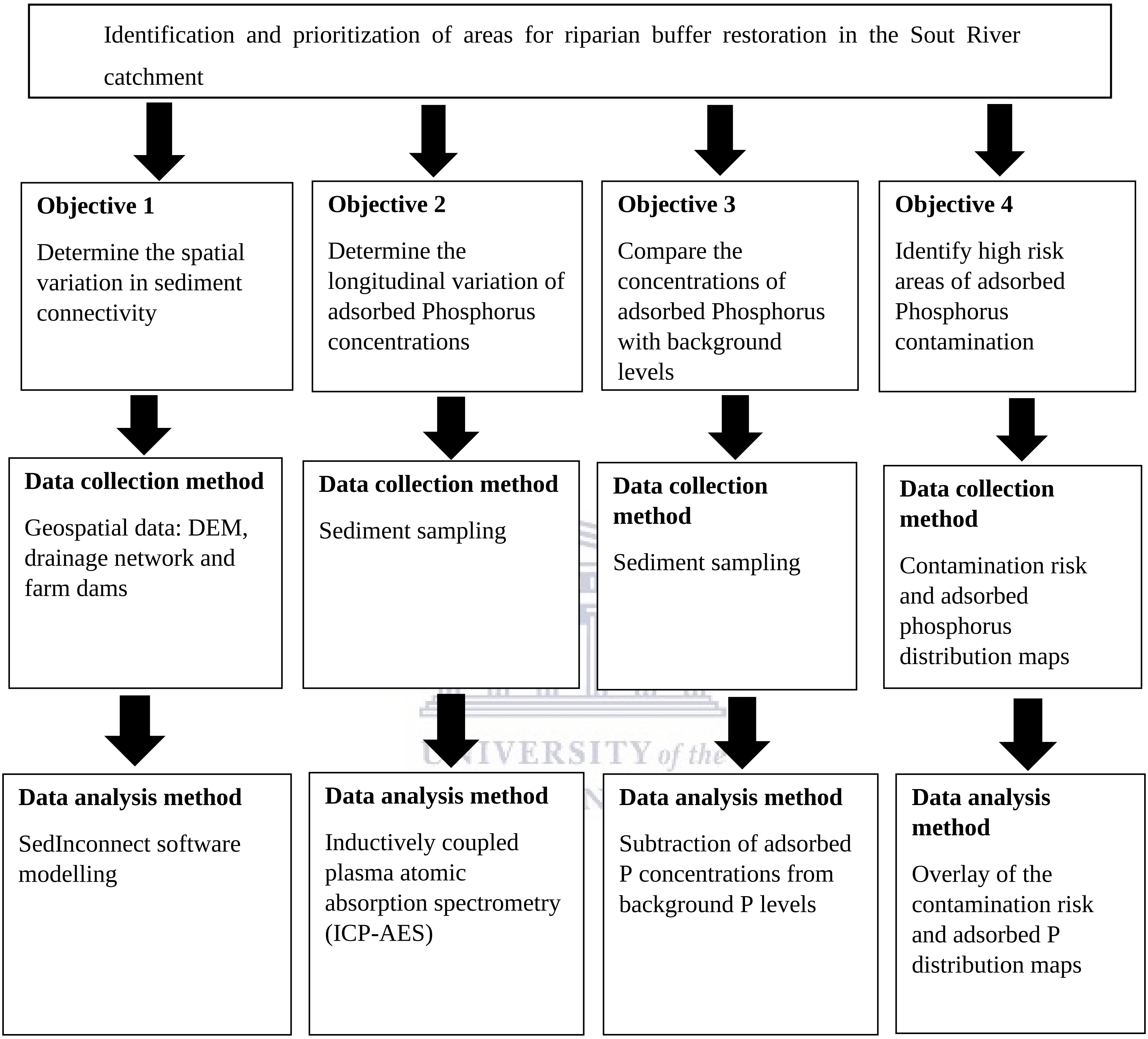


Figure 1.1: Research framework of the present study.

1.8 Outline of the thesis

The thesis outline is as follows: Chapter 1 provides the general background of riparian vegetated buffers and their function in reducing sediment fluxes into watercourses. Additionally, the research question, central argument, aim and objectives, and the significance of the study are also discussed in this chapter. Chapter 2 presents a review of literature and identifies gaps in knowledge regarding the research topic. The geographical area of the study site and broader research region, including the climate, geology, vegetation, drainage and land use information are discussed in Chapter 3. Chapter 4 provides the description of the research design as well as data collection and data analysis methods. Chapter 5 presents and describes the results obtained from the data analysis. The results are interpreted and discussed in Chapter 6. Finally, Chapter 7 provides key findings, study limitations and recommendations for future research.

Chapter 2: Literature review

2.1 Introduction

While riparian buffer restoration has long been recommended as one of the important measures in preventing diffuse pollution from agricultural lands, there is still a lack of a clear strategy for determining areas in the catchment where restoration should be prioritized. The movement of sediments in a landscape is characterized by the spatial and temporal irregularities along the pathways from source areas to the sinks (Calsamaglia et al., 2017). Therefore, to identify areas that should be prioritized for riparian buffer restoration, there is a need to understand sediment pathways and thus sediment connectivity. This chapter will focus on understanding the concept of sediment connectivity and its application in sediment management and will shed light on riparian buffer strips and diffuse pollution caused by phosphorus (P) that comes from agricultural lands. This chapter will also review the South African guidelines on riparian buffer determination and identify a gap in knowledge.

2.2. Previous studies on riparian buffers

Over the past decades, pollution associated with agricultural activities has been recognized as a serious threat to water quality. Certain agricultural practices degrade surface water quality by increasing stream bank erosion which results in contaminants, bacteria, and nutrient loadings (Hickey and Doran, 2004). This growing concern has resulted in the development of best management practices that minimize the impact of these practices. Best management practices describe ways in which non-point source pollution con be controlled.

Riparian vegetated buffers have been advocated for as one of the best management practices in reducing non-point source pollution in watercourses. They are a linear band of natural vegetation adjacent to an aquatic ecosystem. They provide a wider range of ecosystem services in catchments than non-riparian field margins. For instance, in addition to improving the scenic value of the environment and preserving biodiversity, they protect watercourses against diffuse pollution (Dillaha et al., 1989). Properly designed and strategically located in the landscape, they can effectively reduce the movement of sediment nutrients from and within agricultural lands (Dillahha et al., 1989). This is because vegetation increases the hydraulic roughness of runoff, which reduces overland flow velocity and sediment transport capacity (Dillaha et al., 1989). The reduction in flow velocity and the increased resistance to flow promotes infiltration and deposition of sediments. Riparian buffer strips intercept pollutants through a range of physical, hydrological, chemical, and biological processes (Fischer and Fischenich, 2000).

Vought et al. (1995) emphasized that a change in agricultural practices alone cannot solve the nutrient and sediment non-point source pollution of watercourses. It is further added that there is a need to incorporate existing knowledge when developing management methods that are aimed at reducing the risk of such pollution having negative environmental consequences (Vought et al., 1995). Hence, riparian buffers restoration has been suggested by several researchers around the world as a means of reducing diffuse pollution in watercourses (Hawes and Smith, 2005; Richardson et al., 2007; Adams et al., 2020; Cole et al., 2020). From the South African context, Macfarlane et al. (2015) proposed vegetated buffer strips as a standard mitigation approach to reduce the effects of land use activities near water resources.

The effectiveness of riparian buffers in filtering sediments is well studied. Patty et al. (1997) found that riparian buffer strips could reduce 87 to 100% of suspended sediment, 47 to 100% of nitrate, 22 to 89% of soluble P and 44 to 100% of the herbicide atrazine from agricultural runoff. A study by Dillaha et al. (1989) found that a riparian buffer strip removed an average of 84% of suspended solids, 79% of phosphorus, and 73% of nitrogen from cropland. Several other studies also show that riparian buffer strips are effective in reducing pollutants from runoff (e.g., Vought et al., 1994; Syversen, 2002; Uusi-Kämppä et al., 2000).

Buffer characteristics such as width, slope, vegetation cover and type play a significant role in determining riparian buffer effectiveness. Of these, buffer width and slope have been found to be the most significant in determining buffer efficiency (Liu et al., 2008). Extensive research has been conducted to determine the influence of buffer width on nutrient retention (e.g., Uusi-Kämppä et al., 2000; Hawes and Smith, 2005; Zhang et al., 2010; Cole et al., 2020). Although the relationship varied due to site specific conditions (e.g., topography, soil type and vegetation structure, pollutant type), buffer width was found to be positively related to buffer efficiency (Cole et al., 2020). Additionally, Hawes and Smith (2005) indicated that the width of a buffer depends largely on the resource that is being protected. It is then recommended that buffer width is determined based on risk and site-specific conditions (Fischer and Fischenich, 2000; Macfarlane et al., 2015).

Buffer slope is another characteristic that influences buffer effectiveness. A non-linear relationship was observed by Liu et al. (2008) where they found that as the slope increased, buffer efficiency also increased; however, this was only up to a certain point after which steeper slope decreased buffer efficiency. This finding indicated that an optimal slope of 9% is required to achieve maximum buffer efficiency (Liu et al., 2008). In areas with steeper slopes runoff tends to flow through the buffer too fast and thus reducing the buffer's effectiveness in removing sediments. On the other hand, relatively flat slope allows the buffer to readily reduce runoff velocity, increasing infiltration and sediment deposition (Tomer et al., 2003). Therefore, it is necessary to use methods that integrate all these factors in the establishment of vegetated riparian buffers.

The morphological and functional characteristics of plants also influence the effectiveness of vegetated buffers (Fischer and Fischenich, 2000; Cole et al., 2020). Buffers composed of a mix of several types of vegetation structures have been found to be more effective in removing pollutants than buffers that consist of only one type (Fischer and Fischenich, 2000; Jontos, 2004; Hawes and Smith, 2005). Additionally, indigenous vegetation is more preferred as it was found to be more effective in removing contaminants (Fischer and Fischenich, 2000; Hawes and Smith, 2005; Zhang et al., 2010; Petersen et al., 2020).

Due to the extensive range of factors influencing the functions of vegetated buffers, the effectiveness of riparian buffers as a mitigation tool varies depending on site-specific conditions. Identification and prioritization of riparian buffer restoration in a catchment is crucial as it improves the cost-effectiveness of establishing them (Zhao et al., 2013). Targeting areas that release the largest quantities of non-point source contaminants increases buffer effectiveness (Qiu, 2009). Buffer restoration should be prioritized in locations where they are managing more heavily contaminated areas (Zhao et al., 2013). However, the identification of such areas is not easy as it depends on numerous factors. These factors vary depending on site-specific characteristics. For example, sediment movement in mountainous catchments is highly affected by topography whereas it is influenced more by land cover in lowland catchments. As a result, there has been attempts by researchers to develop methods that can easily be used to identify such areas in the different catchment settings. These studies are reviewed on global, regional, and local scales in the sections that follow.

2.2.1 Global context

Riparian buffers efficiency in a catchment differs from one location to another due to site specific characteristics. Various soil and landscape processes influence how water and sediments move across riparian zones towards a channels network. Therefore, it is crucial to re-establish buffers in locations where their benefits will be maximized while putting these processes into consideration. As a result, several methods have been proposed using different approaches to identify buffer priority areas. However, all these studies use different approaches indicating that there is a need for a cost-effective approach that can be applied in any catchment setting.

Spatially distributed models have been developed to assist managers with identifying areas where buffers can have a relatively greater impact on the water quality of a watercourse. Two general approaches have been developed whereby one approach makes use of soil surveys while the other one uses topographic data to identify buffer priority areas. The soil survey method derives numerical indices based on soil map attributes from publicly available data. It identifies map units that are more conducive to contaminant deposition and infiltration in overland runoff as riparian buffer priority areas (Dosskey et al., 2006). In the topography-based method, terrain analysis is used to derive a numerical index from a digital terrain model (DTM) (Dosskey and Qui, 2010).

The soil survey technique uses soil survey attributes such as slope, soil texture and erodibility to determine the soil's capability to remove sediments from runoff. Soil survey data is used to calculate sediment factor for each soil map unit based on Revised Universal Soil Loss Equation (RUSLE) variables (Dosskey, 2008). The sediment factor value is used to estimate the Sediment Trapping Efficiency using a calibration equation, which is determined by employing the Vegetative Filter Strip Model (VFSMOD) (Dosskey, 2008). The sediment trapping efficiency ranks each soil map unit according to its ability to trap sediment contained in surface runoff from agricultural fields (Tomer et al., 2009). A map that highlights where the soil would be more effective at removing sediments and thus, should be prioritized for riparian buffer restoration is then created from this data.

Dosskey et al. (2006) applied this technique in a catchment in the US. They used the soil survey technique to identify areas in the catchment that would remove sediment from surface runoff more efficiently. The sediment trapping efficiency values varied from 21 to 99%, indicating a strong variability across the map units. This variability was due to factors such as rainfall characteristics, soil texture and slope (Dosskey et al., 2006). The results also revealed that higher sediment trapping efficiency values were associated with gentle slopes and lower surface runoff loads. On the other hand, lower sediment efficiency values were found on slopes that are relatively steeper and that generate large runoff loads. This approach was able to show

how buffer efficiency differs from one location to another in the same catchment making buffer placement a very crucial factor to consider during the buffer restoration process.

Dosskey (2008) also applied this method in the Chesapeake Bay catchment in the US and the results were compared with expert opinion to determine the usefulness of the soil survey method in determining the spatial variation in buffer performance in different soil map units. The comparison results found that both methods produced similar results. It was concluded that the soil survey technique is a valid method that can be used to determine the efficiency of soil map units in removing sediments from runoff, and the method can be applied at smaller spatial scales. However, the limitation of this method is that it only considers soil characteristics and slope in determining buffer effectiveness and ignores other factors such as surface roughness that also contribute to the impedance of sediment flow.

Terrain analysis calculates and maps the amount of upslope contributing area that has more potential of delivering overland flows to each grid cell position (Tomer et al., 2003). This data is then used to analyse patterns of overland flow across the landscape (Tomer et al., 2003). In this approach, buffers are prioritized in areas where more runoff water converges from the uplands and saturates the soil (Dosskey and Qui, 2011).

The terrain analysis method was applied by Tomer et al. (2003) to optimize placement of riparian buffers in the glacial terrain of the Des Moines lobe in the US. They analysed a 30 m (digital elevation model) DEM and used it to identify riparian areas with the highest wetness indices where buffer vegetation is likely to intercept sheet flows from substantial upslope areas. This was based on the idea that it would be more beneficial to place a buffer where it will receive runoff from a large upslope contributing region than where it would receive runoff from a small upslope area (Tomer et al., 2003). The study found terrain analysis to be useful in identifying areas where buffers would most effectively improve water quality. However, it was recommended that a DEM with a higher resolution and field observations should be used to improve the results.

Piechnik et al. (2012) used 30 m, 10 m, and 1 m DEM resolutions to investigate the influence of DEM resolution in identifying buffer priority areas to reduce water quality impacts from pastures. They compared the topographic flow path length and stream entry points estimated from the three DEMs with each other and with the Euclidean distance to the stream. Drainage area for each streambank cell produced using these three resolutions were also evaluated. These analyses were done throughout the riparian zone, within the agricultural land use zone and within the identified heavy animal land use zone. It was found that the 30 m DEM was too coarse to provide reasonable flow path estimates in the study area while the 1 m DEM resulted in erratic flow path estimates. The 10 m DEM was found to be more suitable for assessing pasture buffers in this area. This supports the recommendation by Tomer et al. (2003) who recommended that a DEM with a resolution higher than 30 m should improve the terrain analysis results. However, as much as this method is helpful, it is not enough to determine appropriate buffer placement because it only considers the amount of flow an area is likely to receive and does not consider pollutant concentrations in those areas.

These methods have been found useful, however, their application come with certain advantages and limitations. For instance, unlike the topographic method, the soil survey technique does not consider the variation in size of field runoff areas as well as saturation and flooding (Dosskey and Qui, 2010). Additionally, these methods require different input information, therefore, they may potentially lead to different areas being identified. Tomer et al. (2009) compared the two landscape analysis methods and suggested that since they are complementary, they can be combined. Hence, the present study also follows an integration approach.

Qiu (2003) proposed a strategy for placing buffers based on variable source area (VSA) hydrology. Additionally, Qiu (2009) used a modified topographic index approach based on VSA hydrology to identify critical source areas in Neshanic River catchment, USA. The proposed method was found to be a powerful screening tool for identifying sites for riparian buffer placement. When compared to traditional riparian buffer scenarios for placing conservation buffers in agricultural lands, this GIS method was found to be more cost-effective.

Song et al. (2012) developed two riparian forest indices, Riparian Forest Index (RFI) and Riparian Change Forest Index (RCFI), using LANSAT images with 30 m resolution to represent spatiotemporal change of catchment riparian forest to prioritize areas for riparian placements at a sub-basin level in Korea. The results revealed that land cover in the riparian area increased while forests and croplands decreased. Additionally, riparian forest removal occurred more rapidly in the riparian areas. The study found that areas that should be prioritized for riparian placements are those located upstream of the catchments. However, the limitation of this study is that it only focuses on the buffer area and does not consider characteristics of the contaminants. It is important to consider contaminant characteristics when identifying priority areas as they influence buffer efficiency.

To overcome the limitation that previous studies had of only considering one aspect that would affect buffer planning, Zhao et al. (2013) proposed a multi-criterion planning scheme for evaluating riparian buffer priorities using a river basin in China as a case study. This study integrated six indicators, total nitrogen export, total phosphorus export, vegetation vigour, soil erodibillity, mean buffer width, and buffer gap ratio, to assess priority areas. Soil and Water Assessment Tool (SWAT) model was used to simulate total nitrogen and phosphorus exports moving from upslope to the stream while the other indicators were calculated using GIS. The results of the multi-criteria analysis were used to produce a distribution map the ranked the buffer restoration priorities. According to the findings, priority should be given to sub-basins located in the lower sections of the river basin when restoring riparian buffers.

2.2.2 Regional context

In the traditional African context, nature and people are viewed to be the same and as a result, there is no separation between the two (Lelo et al., 2005). For many years people in communal Africa have maintained their way of life in riparian areas without seriously harming the ecosystem (Enanga et al., 2010). However, due to the increasing human population, the intensity of land use change to commercial agriculture has also increased (Enanga et al., 2010). Consequently, it has become essential to manage these disturbances and protect these areas. Currently there are only a few studies that have been conducted on the African continent that focus on the restoration of riparian buffers.

In Kenya, a study was conducted to determine how land use activities affect riparian vegetation along the Njoro and Kamweti Rivers (Koskey at al., 2021). The major land use categories identified on the two sites were forest, agriculture, and build-up areas, and these were used as sampling sites. Results revealed that trees and shrubs were dominant in the forest areas while herbs and shrubs dominated the agricultural and built-up areas along both rivers. It has been found that the decrease in plant species diversities as well as the vegetation composition and distribution across the different land uses can be attributed to anthropogenic activities along both rivers. The study concluded that there is an urgent need for an integrated approach for the management of riparian areas which is what the present study addresses.

In Nigeria, Chukwuka and Ogbeide (2021) conducted a study in the Ikpoba River catchment which focused on riparian buffer-loss and pesticide. The study aimed at using pesticides incidents to demonstrate the suitable riparian buffer width for the protection of surface water, sediment, and benthic fish populations. A normalized difference vegetation index (NDVI) was used to classify riparian areas according to vegetation richness and a multiple-buffer analysis was utilized to determine buffer width. The results revealed that sites with narrower buffer width and sparse riparian vegetation had higher pesticides content compared to sites with wider buffer widths. This indicated that the presence of dense riparian vegetation along the river filters pollutants generated from adjacent land use activities and forms a protective barrier for the stream. It was for this reason that the study concluded that riparian buffer width and density play a crucial role in the filtering capacity of a buffer. The findings of this study validate the urgent need to restore riparian vegetation and highlight some of crucial factors that must be considered in the process.

2.2.3 Local context

The restoration of vegetated riparian buffers as a protective measure against non-point source pollution is also advocated for in South Africa (Macfarlane et al., 2015). However, only a limited number of studies regarding this topic have been done. Earlier studies of riparian buffer restoration include that of Blanche (2002) who focused on proposing guidelines to facilitate recommendations regarding the establishment and management of riparian zones in agricultural landscapes. Additionally, Macfarlane et al. (2015) proposed guidelines for the determination of buffer zones for rivers, wetlands, and estuaries. The main aim of this report was to provide concepts, background and approach required to determine appropriate aquatic buffer zones. In addition to the report, a technical manual and a practical guide were also published (Macfarlane and Bredin, 2017, 2016). The technical manual details the step-by-step procedure used to determine suitable methods for determining suitable buffer zones for rivers, wetlands, and estuaries. It also provides information on the justification for the chosen strategy taken as well as technical information that served as the premise for developing the tools for riparian buffer determination. The main objective of the practical manual is to provide information on the practical application of the buffer zone determination tools. However, these guidelines were developed based on international studies.

More recently, Petersen at al. (2020) studied the effectiveness of riparian vegetation in reducing water quality impacts in an agriculturally dominated system of the Klein Keurbooms River in the Western Cape. The study found that indigenous vegetation cover provided maximum ecosystem services as it was more effective in reducing nutrient and sediment content and therefore recommended, they be used when re-establishing riparian buffers. This study was able to show that the re-establishment of riparian buffers can be applied in agricultural dominated South African catchments to improve water quality.

Studies on riparian buffers are limited in South Africa. Especially those that focus on identifying sites for riparian buffer restoration. The available buffer guidelines only provide information on buffer zone destination and design (Macfarlane et al., 2015). Moreover, these guidelines are based on international studies, which means they may not be representative of local conditions. Therefore, there is an urgent need to invest more in riparian buffer studies. Identification of priority areas for riparian buffer restoration is even more in this country due to limited resources.

2.3 The Sediment connectivity concept

The occurrence of extreme precipitation events results in excessive amounts of runoff flowing at high velocities. This runoff usually has the capacity to erode substantial amounts of sediments from their sources and deposit them at the nearest sinks, which are usually watercourses. In agricultural settings, these sediments usually contain high concentrations of adsorbed phosphorus which can eventually degrade water quality when deposited into streams. This erosion and transport of sediments and soil particles is a natural process that can also be accelerated by anthropogenic activities. Some of these activities include the reduction of soil surface cover provided by vegetation, certain crops, and soil tillage. All these factors influence the movement of sediments from source to sink.

The ease with which sediments move from hillslopes to channel networks depends on geomorphic coupling. According to Harvey (2002) geomorphic coupling is defined as the linkage between the compartment parts of a geomorphic system. Sediments are transmitted easier in well-coupled systems, but they may be spatially constrained in poorly coupled or buffered systems (Harvey, 2002). In this way the structure of the landscape affects the erosion, deposition, and storage of sediments, making it a crucial factor to consider in sediment management studies. In addition to geomorphic coupling, the concept of sediment connectivity also plays a major crucial role in the movement of sediments in a landscape. Sediment connectivity is defined as the degree to which a geomorphic system promotes sediment movement through the spatial organization of geomorphic features and processes (Heckmann et al., 2018).

The sediment connectivity concept is used to describe the flow of sediments from a source to a sink in a catchment, as well as movement of sediment between distinct zones within a catchment: over hillslopes, between hillslopes and channels, and within channels (Bracken et al., 2015). Therefore, sediment connectivity is a key factor in determining the total amount of sediment that reaches a catchment outflow (Masselink et al., 2016). Nutrient laden sediments from hillslopes can have a negative impact on the water quality if they get deposited in streams. Understanding the spatial and temporal variation of sediment connectivity is useful when dealing with sediment management, hazard assessment and planning and design of structural measures (Masselink et al., 2016; Crema and Cavalli, 2018).

2.3.1 Factors affecting sediment transfer processes in a catchment

Natural factors that affect sediment transfer processes include topography, surface roughness, and spatiotemporal dynamics of vegetation cover, effective rainfall, stream network density, soil permeability and water retention (Roehl, 1962; Cammeraat, 2002; Foerster at al., 2014; Mishra et al., 2019). Generally, slopes are potential sediment transport pathways (Bracken et al., 2015), therefore, they promote sediment transport. Vegetation influences surface roughness and local capacity to store sediments and water (Borselli et al., 2008). It increases surface roughness and infiltration, thus decreasing erosion and the transfer of sediments (Sandercock and Hooke, 2011). The reduced transport capacity results in sediment deposition (Poeppl et al., 2012). In this way vegetation plays a role in dis(connecting) landscape compartments such that severely eroded areas in the landscape are likely to be highly connected to the drainage network than densely vegetated landscapes (Borselli et al., 2008). For this reason, vegetation is considered as one of the crucial internal factors that influences the magnitude of erosion and sediment delivery to the drainage network (Poeppl et al., 2012). However, it must be noted that the effects of vegetation on sediment connectivity in a landscape vary greatly according to spatial and temporal dynamics.

Several land use and land cover changes (LULC) caused by anthropogenic activities also influence sediment connectivity and have led to the transformation of the natural sediment transfer processes in most catchments. These include, afforestation, deforestation, agriculture, build-up areas, clearing of riparian vegetation (Mishra et al., 2019). These LULC changes can either promote sediment transport processes or impede sediment transfer in a landscape. Generally, LULC such as deforestation and agriculture can accelerate sediment transfer processes due to increased soil erosion and increased transfer rates from sediment source areas to the drainage network (Poeppl et al., 2012). On the other hand, afforestation and build-up areas can reduce sediment transfer by acting as buffers.

In addition to catchment characteristics, sediment properties such as density and size also influence how sediments get transported in a landscape. For example, earlier research has revealed an inverse relationship between sediment particle size and sediment-transport distances (Wainwright and Thornes, 1990). This means that larger sediment particles get deposited over short distances whereas smaller particles can travel long distances before they can be deposited. This makes it important to consider sediment properties when managing sediments in a catchment.

Considering the above, it is crucial to use a method that accounts for these factors when estimating sediment connectivity in catchments. Research in sediment connectivity and the general sediment transfer processes has expanded over the years, more methods have been introduced to estimate this phenomenon. However, Poeppl et al. (2012) noted that there is lack of knowledge when it comes to the effects of riparian vegetation on sediment connectivity as well as on the processes and factors that govern them. There is a need to incorporate sediment connectivity assessments in riparian buffer restoration plans and this presents an important research gap.

2.3.2 Sediment connectivity assessment methods

Sediment connectivity cannot be explicitly measured (Turnbull et al., 2018). However, to gain a better understanding of landscape sediment transport processes and to develop effective sediment management strategies, several techniques have been developed to quantify sediment connectivity. These techniques include indices (Borselli et al., 2008; Hooke and Sandercock, 2012; Gao and Zhang, 2016; D'Haen et al., 2013; Zanandrea et al., 2020), models (Messenzehl et al., 2014; Schmitt et al., 2016; Di Stefano and Ferro, 2018; Mahoney et al., 2018) and graph theory (Heckmann and Schwanghart, 2013; Cossart and Fressard, 2017; Fressard and Cossart, 2019). Among these, indices have been mostly applied in connectivity studies.

A comprehensive analysis by Najafi et al. (2021) revealed that considerable amount of studies have been done to assess sediment connectivity using these main indices (i) sediment delivery ratio (SDR) and sediment yield (Brierley et al., 2006; Rommens et al., 2006; Hardy et al., 2010; Coulthard et al., 2016; Gao and Zhang, 2016; Masselink et al., 2016; Heckmann and Vericat, 2018), (ii) the index of connectivity (IC) (Borselli et al., 2008; Sougnez et al., 2011; Hooke and Sandercock, 2012; D'Haen et al, 2013; Lopez-Vincente et al., 2013; Foerster et al., 2014; Broeckx et al., 2016), (iii) the modified IC (Cavalli et al., 2013; Gay et al., 2016; Tiranti et al., 2016; Calsamiglia et al., 2018a; Calsamiglia et al., 2018b; Persichillo et al., 2018; Kalantari et al., 2017; Schopper et al., 2019), and (iv) a combination of SDR with IC or erosion with sediment deposition values across the study site (Jamshidi et al., 2014; Hamel et al., 2015; Bywater-Reyes et al., 2017; Prosdocimi et al., 2017).

The SDR has been proposed to measure the magnitude of connectivity in geomorphic systems. It is defined as a measure of the percentage of total catchment erosion transported from the basin (Brierley et al., 2006). As a sediment connectivity indicator, it is interpreted as a reflection of the degree of linkage between the river system and land surface sediment sources (Najafi et al., 2021). However, SDR represents a 'black box' model and therefore, does not provide further information regarding the processes that occur between sources of sediments, sinks and their connection to the catchment outlet (Bracken and Croke, 2007; Hoffmann, 2015). This limits the application of this method in sediment connectivity assessments as these assessments should provide insight into how the sediment conveyor belt functions at a catchment scale and address some uncertainties of the main sediment transportation (Najafi et al., 2021). Additionally, this method does not provide information regarding the amount of sediment contributions from the different areas in the catchment, therefore, more distributed approaches needed to be developed.

The Index of Connectivity (IC) is a GIS-based index proposed by Borselli et al. (2008) which quantifies connectivity based on the upslope and downslope components of the landscape using topography data derived from DEM or DTM and on land use. According to Borselli et al. (2008), IC evaluates the potential of connection between sediment source areas (hillslopes) and areas acting as targets or sediment sinks for transported sediment. Combined with sediment availability information, IC can be used to identify primary sediment sources that have the potential to be transferred downstream as it provides information on links between sediment source and sink areas, making it more advantageous to use compared to SDR (Najafi et al., 2021). Due to its simplicity and limited data requirements, IC can be readily applied at a catchment scale, and this makes it an attractive tool for researchers doing sediment connectivity assessments (Gay et al., 2016).

This method has been widely accepted and applied in many sediment connectivity studies around the world and has shown to be efficient in diverse geographical environments. For example, it has been used to monitor contaminated sediment dispersion in Japan (Chartin et al., 2013), to improve sediment yield prediction in a semi-lumped catchment model in south-east Australia (Vigiak et al., 2012), and was applied to investigate the effectiveness of vegetation in reducing desertification and land degradation in Spain (Hooke and Sandercock, 2012). The many applications of this index have proven its usefulness in sediment connectivity studies. However, just like any other index, it also comes with limitations that must be taken into consideration when applying it.

Borselli et al. (2008) used the C-Factor from USLE-RUSLE as the weighting factor (W) which is used to model the resistance to runoff and sediment fluxes caused by characteristics of local land use and soil surface when estimating IC. The C-Factor is used to determine the relative effectiveness of crop and soil management systems in terms of preventing soil erosion (Borselli et al., 2008). It attains its maximum value when the soil is at the highest risk of erosion and goes close to zero when the soil is more protected (Borselli et al., 2008). The authors did emphasize that it is also important to derive W from surface features that affect runoff processes and sediment fluxes within a catchment or on a hillslope. For this reason, Cavalli et al. (2013) proposed modifications to the index by replacing the C-Factor with the roughness index (RI) which is derivable from the digital terrain model (DTM). Additional modifications to the index include the use of the multiple D-infinity approach instead of the single-flow direction algorithm, and the use of a threshold in the computation of slope to avoid bias in very steep slopes (Cavalli et al., 2013). These modifications enable the analysis of various sediment transport systems including those that may not be controlled by hydrologic factors (Bracken et al., 2015). Additionally, these refinements enable the model to be applied in mountainous catchments and to use high resolution digital terrain models (HR-DTM) (Cavalli et al., 2013).

This modified version of IC has since been accepted and applied in different environments. For example, it has been used to characterize sediment supply from landslides to the channel network (Tiranti et al., 2016) and to model post-fire sediment connectivity in a catchment (Grenfell et al., 2022). Cavalli et al. (2013) successfully applied it in Alpine catchments where the dominating transfer processes include debris flow and channelized sediment transport.

Topographic indices are based on the idea that the slope steepness and direction determine where the sediment flows (Gay et al., 2016). Hence, IC ignores factors other than topography that can regulate (dis)connectivity in an area. As a result, it may overestimate hotspots of connectivity in lowland areas therefore, its application in lowland areas must be accompanied by either index modifications or field campaigns to assess impedance based on land use (Michalek et al., 2021). For this purpose, Gay et al., 2016 introduced a revised IC that takes landscape infiltration and saturation qualities into account. Kalantari et al. (2017) proposed the evaluation of sediment connectivity based on surface runoff. The present study uses IC to identify areas that should be prioritized for riparian buffer restoration; therefore, the resulting IC is incorporated with vegetation cover to accurately identify these areas.

2.3.3 The application of sediment connectivity assessment

Connectivity varies in both space and time and therefore, the spatial distribution and temporal evolution of sediment connectivity patterns in the catchment may be used to estimate how a given part of the catchments contributes as a sediment source and may also be used to identify sediment transfer paths (Borselli et al., 2008; Cavalli et al., 2013). The identification of sediment source areas and their connection to the channel network is important for environmental management (Foerster et al., 2014). Sediment connectivity assessments have been done in catchments all over the world to achieve various objectives. Najafi et al. (2021) emphasized the importance of including connectivity in management concepts, particularly in developing countries where there are high erosion and sediment delivery rates.

Sediment connectivity assessment is a useful tool that should be incorporated in sediment management strategies such as riparian buffer restoration. However, this has rarely been done before. The assessment of sediment connectivity in a landscape can help with identifying sediment source areas and pathways, and thus making it easier to identify points that should be prioritized for riparian buffer restoration. According to Najafi et al. (2021), sediment connectivity assessment may be used for spatial prioritization as an effective screening tool for sediment management in catchments. A landscape that is highly connected to the watercourse is more likely to enhance sediment transfer and thus promote diffuse pollution. This means that to address the issue of diffuse sediment pollution measures should be put in place to reduce sediment connectivity, particularly near watercourses.

The efficiency of a riparian zone to buffer sediments flux from agricultural areas to the drainage network is highly dependent on its vegetation cover due to the factors discussed in the previous section (Poeppl et al., 2012). As a result, riparian vegetation may disconnect sediment sources on the hillslopes from the river channel. Poeppl et al. (2012) studied the effects of riparian vegetation on diffuse lateral sediment connectivity using a geomorphic field survey, GIS-based overland flow pathway modelling as well as multivariate statistics and their results revealed that indeed lateral sediment connectivity is influenced by riparian vegetation as forest vegetation significantly reduced sediment inputs into the river channel.

2.4 Phosphorus in soils

Phosphorus is a naturally occurring, essential macronutrient required for plant growth and other crucial plant functions. It plays a crucial role in numerous physiological and biochemical processes and cannot be replaced by any other element. Phosphorus deficiency in soils can reduce plant growth and development, and potentially limit crop yield. It is a naturally occurring element found in rocks, soils, and organic matter. It has been estimated that 30 to 65% of total soil phosphorus is in organic form while 35 to 70% is in inorganic form (Rishid, 2019). Organic forms of phosphorus include decaying plant material and soil microorganisms. Plants absorb inorganic orthophosphates, $H_2PO_4^-$ and HPO_4^{2-}, but can also absorb other forms of organic phosphorus (Syers et al, 2008).

Despite being abundant in soil in both organic and inorganic forms, phosphorus is a limiting factor for plant growth (Harding and Paxton, 2001). This is because the P in soils is not readily available to be taken up by plants as it is 'fixed' to soil particles and sediments. As a result, manure and fertilizers containing P along with other essential nutrients are often applied to enhance plant growth and increase crop production. The P used in fertilizers is obtained by mining deposits of the phosphate rock which is then processed, formulated into fertilizer, and applied to the soil.

The addition of P to the soil system alters the solid solution equilibrium, which is dependent on time, the concentration of the different forms of P in the solution, and the soil properties (Dorioz et al., 2006). The soil's ability to control this equilibrium is referred to as its fixation capacity. A higher fixation capacity results in a higher efficiency of P uptake and the quantity of incoming P stored in the soil (Dorioz et al., 2006). P is 'fixed' through sorption to mineral phases and transformed into molecules with less bioavailability (Stutter et al., 2015). Phosphorus adsorbs to the soil particles by displacing other anions with lower affinity. The fixation capacity in agricultural soils is generally large and ideal agronomic levels of solution P may not be maintained even for one agricultural cycle (Menezes-Blackburn et al., 2018). As a result, phosphorus fertilizers are usually applied more than what plants need to overcome P soil fixation processes and keep soil solution P at an ideal level for plant growth (Syers et al., 2008; Stutter et al., 2015). However, only about 10-30% of this phosphorus is utilized by plants (Gupta et al., 2020). This has resulted in the accumulation of limited bioavailable P in soils of many agricultural landscapes which end up being eroded as sediments (Carpenter, 1998).

Although it is clearly beneficial to add phosphorus to the soil, "over-applications" can have detrimental effects on the water quality of catchments that receive P that is lost from agricultural soil through runoff and/or erosion (Poswa, 2016). This results in conflict between agriculture and good ecological state of watercourses: high concentrations of nutrients are desired in agriculture to enhance productivity whereas moderate levels are desired in watercourses to maintain relatively low productivity (Laakso, 2017).

2.4.1 Catchment characteristics influencing phosphorus distribution

To assess the risk of phosphorus contamination in watercourse it is crucial to not only assess its concentration in the sediments but to also determine its spatial distribution throughout the catchment. Catchment characteristics such as topography of the landscape, farming practices and land use influence the distribution of phosphorus (Adhikari et al., 2021). Contradicting results have been found when it comes to the influence of topography on soil P distribution. For example, terrain attributes were used to map soil P distribution in four different fields in North-eastern China and found that elevation and slope had positive correlations with Soil P in two fields, as well as negative correlations in two other fields (Shen et al., 2019). Mage and Porder (2013) and Adhikari at al. (2018) reported a positive relationship between soil P and slope. More recently, an analysis by Adhikari et al. (2021) linked topographic relationships with soil P distribution and concluded that this link exists because topography controls water flow and distribution and influences farm management practices. Meanwhile, Wang et al. (2009), Cheng at al. (2016) and Xu et al. (2014) found that soil P was negatively related to slope and elevation. It can then be concluded that the variation in topographical influence on soil P distribution differs in different areas due to differences in soil age and development, climatic conditions, and topographic heterogeneity (Li et al., 2016; Wang et al., 2010; García-Velázquez et al., 2020).

Land use and land cover type is also another factor that influences P distribution in a catchment. For instance, agricultural land cover is typically associated with activities such as tillage, which increase soil erosion and thus, increased P loss (Allafta et al., 2020). Furthermore, the application of fertilizers and manures in agricultural lands alters the natural P concentrations and if applied in excess can ultimately accumulate in soils resulting in an unequal distribution. Moreover, a positive correlation has been reported between P content and urban land use whereas forest area ratio was found to be negatively correlated with P content (Brett at al., 2005; Ide et al., 2019). However, contradicting results were obtained by Allafta et al. (2020)

who did not find any correlation between P concentrations and urban land cover. Cheng at al. (2016) found that soil P concentrations were the highest on a forestland, followed by cropland and were the lowest at a grassland.

2.4.2 Soil/sediment characteristics influencing soil P adsorption

Inorganic soil phosphorus can be classified into three pools: soil solution phosphorus, sorbed phosphorus and mineral phosphorus. The overall distribution of phosphorus is controlled by a series of physical, chemical, and biological processes which regulate the exchange dynamics among these pools (Dorioz et al., 2006). The main processes are sorption-desorption and dissolution-precipitation. Sorption reactions are relatively fast and reversible surface reactions (adsorption) whereas desorption reactions are the opposite at common soil solution P concentrations (Dorioz et al., 2006; Menezes-Blackburn et al., 2018). P can permeate into the matrix of the adsorbent substrates after being adsorbed, strengthening the chemical bonds (Dorioz et al., 2006). Sorption is a term used to characterize all the processes resulting in the removal of phosphate from soil solution by adsorption and precipitation processes (Asomaning, 2020). These processes determine P concentration in sediments.

Studies have shown that the main soil characteristics that influence P adsorption are pH, temperature, and sediment composition such as organic matter content, clay type and content, as well as the concentration of exchangeable aluminium (Al), iron (Fe) and calcium (Ca) (Hansen et al., 2002, Wang et al., 2006; Wang et al., 2009). Sediment composition is an essential part of phosphorus pollution studies. Organic matter content and particle size are often used to explain the concentrations of phosphorus adsorbed onto sediments in sediment-based pollution studies (Fredrick, 2001). Numerous earlier studies showed that soils and sediments with different particle-size had varying chemical compositions and levels of stability to microbial degradation, and as a result, had different capacities for phosphate sorption (Wang et al., 2006).

It has been reported in several studies that P adsorption capacity is high in soils with a high clay content due to the larger specific surface area compared to sandy soils (Raats et al., 1982; Holtan et al., 1998; Gérard, 2016; Hansen et al., 2002; Gupta et al., 2020). The specific surface area on clay particles determines the availability of sites on the solid phase to take up P (Dorioz et al., 2006). This positive correlation is also due to iron and aluminium oxides on the surfaces of the clay particles (Fredrick, 2001; Wang et al., 2006; Asomaning, 2020). Therefore, sediments with a higher clay content as well as high concentrations of Fe and Al-oxides are likely to have higher P concentrations. It is in such an environment that can potentially become a source of adsorbed phosphorus.

Most of the phosphorus in soils is either adsorbed into organic matter or incorporated into soil particles (Holtan et al., 1988). Hence, organic matter content is related to phosphorus concentrations in sediments. The impact of organic matter content on P adsorption is ambiguous (Holtan et al., 1998; Asomaning, 2020). This is because organic matter inhibits crystallization of iron oxide thereby increasing P sorption and it also competes for adsorption sites and therefore decreases P sorption (Asomaning, 2020). As a result, studies have found contrasting results when it comes to the impact of organic matter on P sorption. Some studies have reported a positive correlation while others did not find any correlation between the two. For example, organic matter did not have direct impact on adsorption of phosphate by Al and Fe oxides in sandy soils (Borggaard, 1986) while Debicka et al. (2016) found that removing organic matter decreased the amount of bound P present in topsoil that was tested. This is because the removal of organic matter resulted in a decreased in P adsorption capacity and an increase in P desorption (Debicka et al., 2016). Another study found that the removal of organic matter increased adsorbed P concentration which shows that organic matter could compete for adsorption thereby inhibiting P adsorption (Hiradate and Uchida, 2004). These contradictory results could be attributed to the type of organic matter and soil type as these are some of the factors influencing P adsorption capacity in soils.

The spatial distribution of P is unique in every catchment depending on catchment and soil characteristics. These characteristics can affect each area differently, resulting in different amounts of P that gets adsorbed to the soil particles. It is therefore crucial to study all these characteristics very closely when managing sediment/nutrient in a landscape and not make assumption based on studies that have been done in other catchments. The spatial distribution of P is useful in determining areas that are more susceptible to P loadings in the catchment.

2.4.4 Phosphorus in watercourses

Phosphorus occurs in natural waters as orthophosphate, polyphosphates, metaphosphate, pyrophosphate, and organically bound phosphate. Only the orthophosphate species, H_2PO_4 and HPO_4^{2-}, can be used by aquatic biota (DWAF, 1996). The exchange of phosphorus between sedimentary and aquatic compartments strongly influences phosphorus cycle in water. Some of the natural factors influencing phosphorus concentration in water are pH, rock type, sorption-processes, and activities of the living organisms. Natural sources of P in aquatic systems include weathering of rocks and decomposition of organic matter. These factors coupled with anthropogenic activities and the flow regimes result in high spatial variation of phosphorus in aquatic environments.

Sediments found in watercourses located in agricultural catchments are usually composed of topsoil of cultivated land which was eroded by runoff (Laakso, 2017). Most of the P is transported from soil to water in sediments enriched with P or through runoff when fertilizers and manures have just been applied to the soil (Syers et al, 2008). This sediment usually becomes the source of phosphorus in watercourses. Under conditions of high flow and anoxic conditions from water and sediments adsorbed phosphorus is released (DWAF, 1996). Nonetheless, phosphorus is rarely found in significant concentrations in South African natural surface waters because of the extensive plant uptake (DWAF, 1996). Concentrations typically vary between 0.01 mg/L and 0.05 mg/L (DWAF, 1996). However, lower concentrations of 0.001 mg/L have also been observed in some 'pristine' waters while higher concentrations of 0.2 mg/L total phosphorus have been found in saline waters (DWAF, 1996).

According to the South African Water Quality Guidelines for aquatic ecosystems, inorganic phosphorus concentration of less than 0.005 mg/kg is thought to be low enough to reduce the possibility of algal growth (DWAF, 1996). Therefore, concentrations that are higher than this are associated with changes in trophic status, as well as growth of algae and other aquatic plants in aquatic ecosystems (DWAF, 1996). Inorganic phosphorus concentrations in all South African surface waters should not differ by more than 15% from those found in the water body under local, unaffected conditions (DWAF, 1996).

2.5 Comparison of adsorbed P to background levels

The sediment background approach compares sediment contaminant concentrations to concentrations from a natural 'reference' site in the same area. Data from reference sites offer a potential baseline for determining the magnitude of change in nutrient levels that has occurred from the time the catchment become loaded with nutrients (Smith et al., 2003). Comparison of adsorbed P concentrations to background levels assumes that concentrations that are not higher than the background levels are not harmful to the environment, thus sediment concentration guidelines are set according to the concentrations of the natural 'reference' site (Persaud et al., 1993). This approach is seen as a simple screening method which requires minimal data (Persaud et al., 1993; Jones et al., 1997). However, it does not consider biological data, and therefore has no biological basis. Furthermore, measured concentrations above background levels are not always hazardous (Gordon and Muller, 2010). Therefore, the approach must be applied with these limitations in mind.

Gordon and Muller (2010) reported that there were no published freshwater sediment quality guidelines (SQG) for freshwater ecosystems in South Africa as they were still under development. As a result, background comparison is often based on international guidelines. SQG that represent the South African aquatic ecosystems are therefore necessary, especially with the increasing agricultural activities which result in loading of contaminated sediments into surrounding watercourses.

2.6 Conclusion

This review has shown the importance of selecting the right areas for the placement of riparian buffers to achieve optimum buffer efficiency. While there are studies that have been done to propose strategies for prioritising areas for riparian buffer restoration, they still fail to integrate the most crucial factors that must be considered when identifying buffer priority areas. Most of these strategies only focus on one factor and do not consider other crucial factors that affect buffer effectiveness, especially the spatial distribution of the pollutant concentration. This presents an important research gap. Buffer effectiveness and opportunity are the main factors that need to be considered when identifying priority areas. In this case, buffer effectiveness refers to sediment connectivity while opportunity refers to adsorbed phosphorus concentrations. Therefore, there is a need to develop an integrated approach that incorporates both these factors in the identification and prioritization areas for riparian buffer restoration.

Chapter 3: Study area

3.1 Introduction

This chapter provides a description of the area where the study was conducted. Description of the study area is given in terms of its physiographic characteristics to understand their influence on sediment movement and adsorbed phosphorus concentrations in the catchment. This chapter focuses on describing physiographic features such as the local climate, topography and land use, vegetation, drainage pattern and the geology of the catchment.

3.2 Study area description

3.2.1 Location

The Sout River catchment is located at 34° 17' 29.976"S and 20° 1' 21.972"E in the Overberg District, Western Cape province, South Africa (Figure 3.1). The catchment lies in the G50 tertiary catchment within the larger Breede-Gouritz Water Management Area. The Sout River catchment comprises mainly two quaternary catchments namely, G50G and G50H. It covers an area of approximately 1200 km^2.

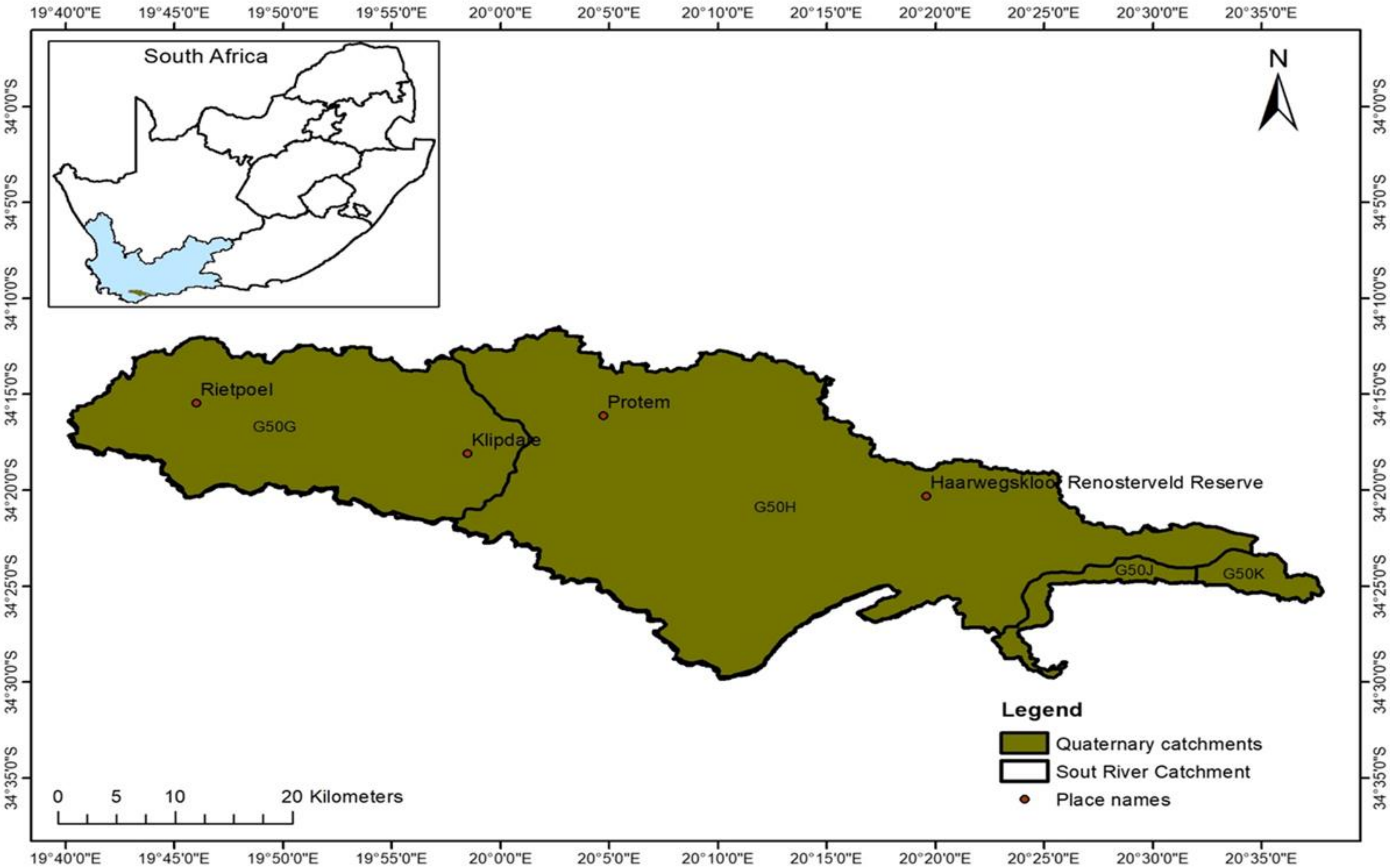


Figure 3.1: Location of the Sout River catchment with respect to South Africa.

3.2.2 Geology and soils

The geology of the Sout River catchment consists of the Bokkeveld Group, the Bidouw and Bredasdorp subgroups as well as small patches of the Nardouw and Ceres subgroups (Figure 3.2). The Bokkeveld Group comprises a succession of mudstones, siltstones, and fine-grained sandstone units. In the Sout River catchment, The Bokkeveld Group occurs south-east of Klipdale in the middle reaches of the catchment through to the north-west of De Hoop Vlei. The Bokkeveld Group occupies the largest area in the catchment. The Bidouw and Bredasdorp subgroups are two of the three subgroups that form the Bokkeveld Group, and they occur at the uppermost and lowermost parts of the catchment, respectively. The underlying Bokkeveld shales in this catchment result in deep clay and loamy soils (Curtis, 2013). The soils are fine-grained and moderately fertile, making this area significant for agriculture (Cowling et al., 1986; Curtis-Scott, 2020).

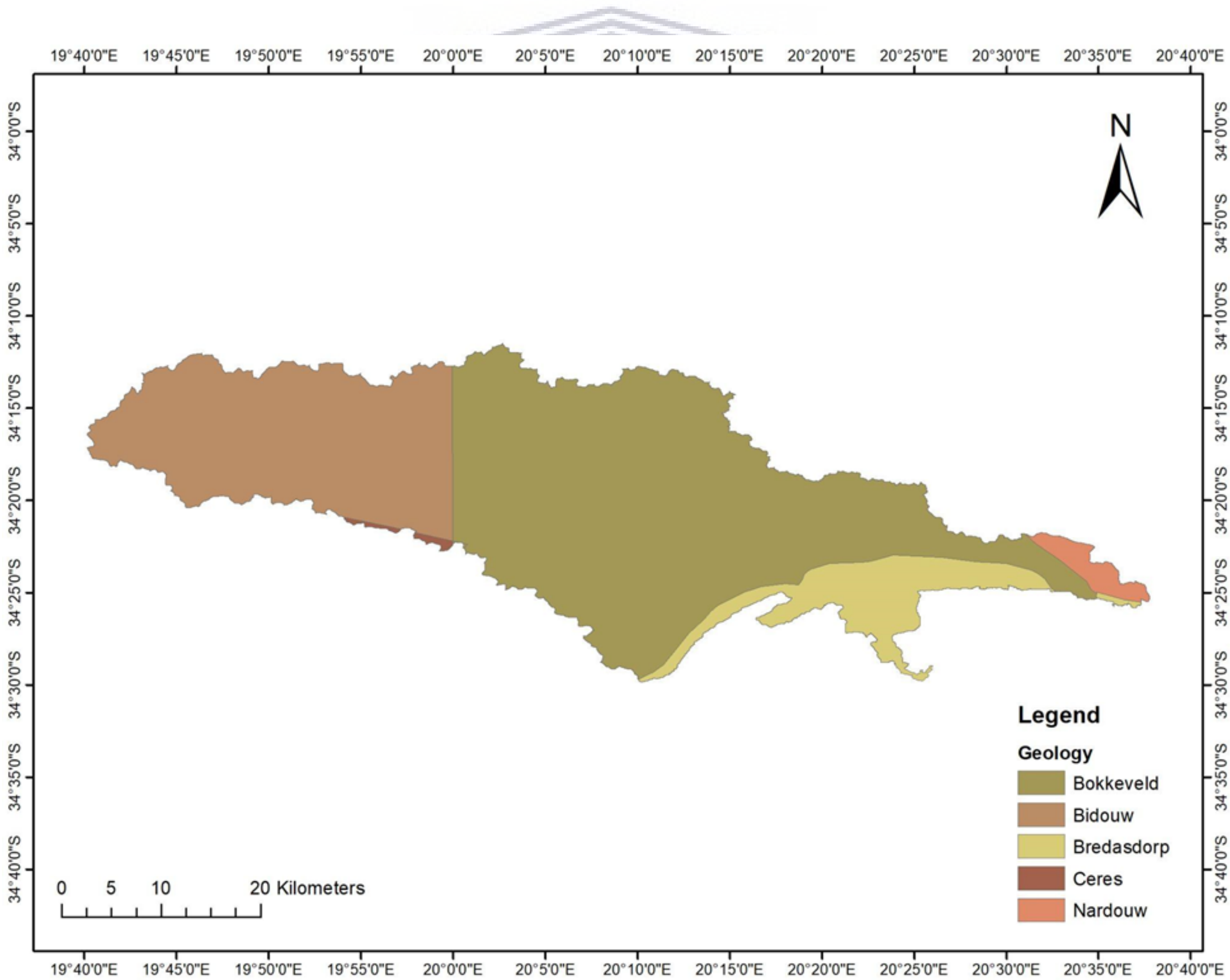


Figure 3.2: The geology of the Sout River catchment.

3.2.3 Vegetation

Lowland Renosterveld is the fertile clay-based vegetation type that is found in this part of the Overberg. This is a vegetation complex that forms part of the Fynbos Biome of the Greater Cape Floristic Region (GCFR), which is popularly known as the smallest yet richest plant region on Earth. Renosterveld vegetation consists mainly of grasses and shrubs of low to medium height (0.5 - 1.2m) which belong to the Daisy family (Curtis-Scott et al., 2020). The Western, Central and Eastern Rûens Shale Renosterveld dominate renosterveld vegetation in the region (Rebelo et al., 2006), while smaller 'islands' of Rûens Silcrete are mostly concentrated within the Eastern Rûens Shale Renosterveld (Curtis-Scott et al., 2020). Rebelo et al. (2006) describes the renosterveld of the Overberg as small-leaved, low to moderately tall grassy shrubland with open to moderately thick cupressoid vegetation that is typically dominated by renosterbos. Figure 3.3 shows the various renosterveld vegetation types that naturally occurred in the Sout River catchment prior to extensive transformation by agriculture.

The dominating vegetation type in the Sout River catchment is the Central Rûens Shale Renosterveld. This vegetation type covers the catchment area from upstream to the middle reaches of the catchment. This vegetation type is distinguished from the other Renosterveld vegetation types in the region by the absence of *Aloe ferox, Pteronia incana* and *Galenia africana* (Rebelo et al., 2006; Curtis, 2013). When compared to Eastern Rûens, Curtis (2013) found that the Central Rûens Shale Renosterveld tends to be grassier and richer in geophyte species.

The Eastern Rûens Shale Renosterveld is the second largest vegetation type in this catchment. It occurs from the middle reaches of the catchment down to the mouth of De Hoop Vlei. Compared to the Western and Central Rûens Shale Renosterveld, it has fewer grasses, however it becomes grassier near the Langeberg foothills, where *Themeda triandra* becomes more dominant (Curtis, 2013). Quartz-silcrete patches, which are recognized for their rich succulent flora and exceptionally elevated levels of plant endemism, are another distinctive feature of this vegetation type (Curtis, 2013). Due to its extremely diverse vegetation, it has been suggested that there might be a need to divide it into several different renosterveld types (Curtis, 2013).

The Western Rûens Shale Renosterveld occurs at the uppermost part of the catchment. According to Rebelo (2006), the absence of *Hermannia flammea* and the rare occurrence of

Aloe Ferrox and *Acacia karoo* distinguishes this vegetation type from other Rûens renosterveld vegetation types. It is typically grassy and has abundant geophytes (Curtis, 2013).

Due to extensive transformation by agriculture, it is estimated that only about 5% of the original extent of Rûens renosterveld remains today (Curtis-Scott et al., 2020). As a result, Rûens renosterveld is listed as Critically Endangered and is highly susceptible to functional extinction (Curtis, 2013). Rûens renosterveld has become highly fragmented, and its remnants are found on slopes that are too steep or rocky to plough and along watercourses and drainage lines (Curtis-Scott et al., 2020).

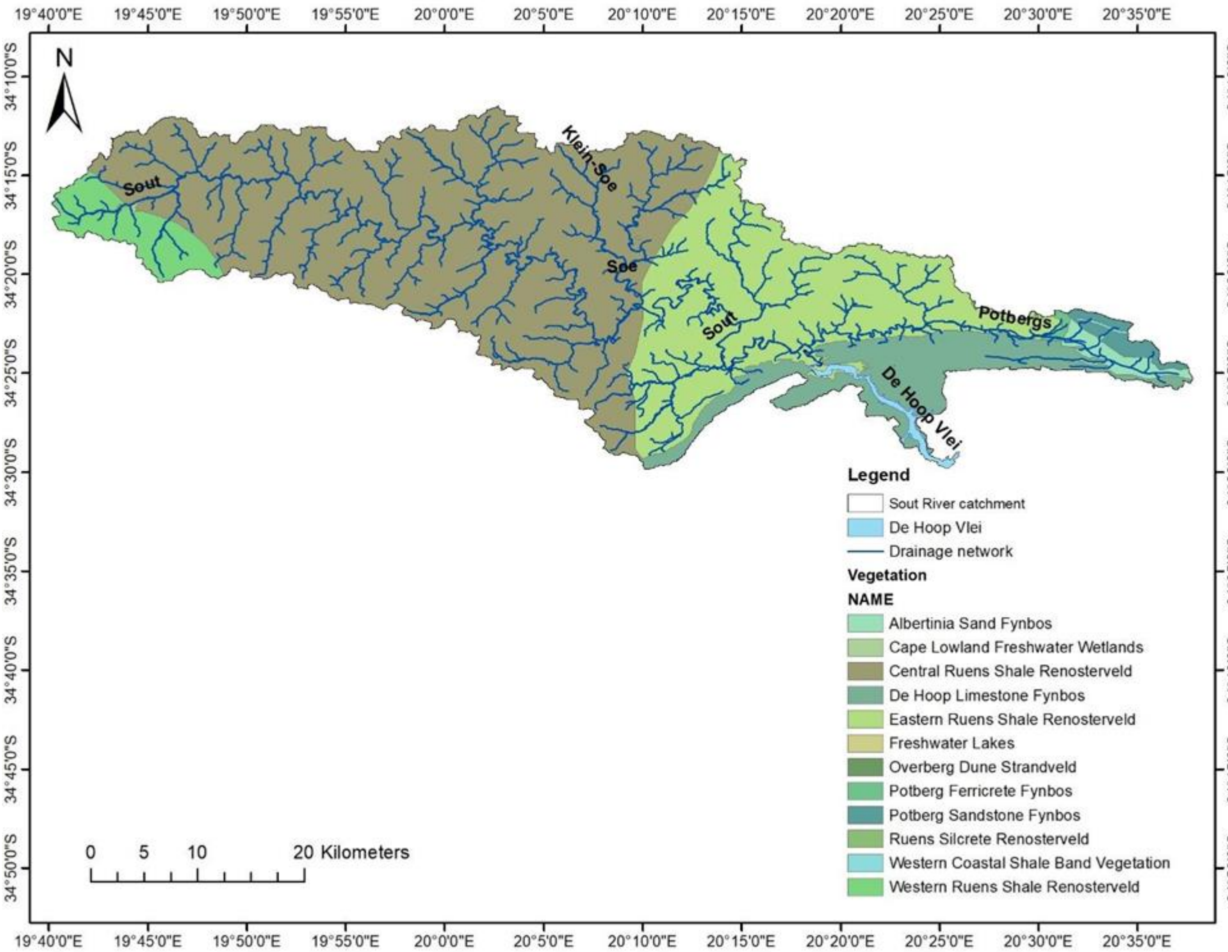


Figure 3.3: Vegetation types in the Sout River catchment.

3.2.4 Topography and land use

Plains with low to moderate relief, and lowlands make up most of the terrain morphology of the catchment (Herdien et al., 2005). The highest elevation in the catchment is 610 m amsl while the lowest elevation is 8 m amsl (Figure 4.3 in chapter 4). The catchment is predominately rural with agricultural activities dominating the landscape. Most of the catchment area is used for livestock grazing and crop farming. It has a varied range of commodities, including livestock, sheep, viticulture, fruit, grain, teas, vegetables, pig farming and crops such as wheat, barley, and canola seed oil (Jepthas and Swanepoel, 2019). This low-lying topography allows agricultural activities to take place right up to the river courses resulting in the modification of the riparian zone (Herdien et al., 2005). Farming encroachment, over abstraction of water, water quality modification, and physical habitat modification are all effects of agricultural practice in this area (Herdien et al., 2005).

3.2.5 Local climate

The region experiences a Mediterranean climate which is characterized by cool, rainy winters and warm, dry summers. The mean daily minimum and maximum temperatures range between 5.6°C and 27.3°C (Rebelo et al., 2006). The mean annual precipitation in the catchment ranges between 300-600 mm and normally peaks in August (winter) (Rebelo et al., 2006).

3.2.5 Sout River hydrology

The Sout River catchment has the Sout River as its major river (Figure 3.2). This is a 141 km long river with a low channel gradient throughout (Lanz, 1997). The Sout River runs across the catchment in a south-easterly direction. Some of the main tributaries of the Sout River are the Soes, Klien-Soe, Waterskilpadsrivier and Potberg Rivers. Most of the channel types are alluvial (Herdien et al., 2005). The channels are dominated by either sand or clay and silt with small percentages of gravel, making the bed reach flat (Herdien et al., 2005). Sout River is an endorheic river and drains into De Hoop Vlei at De Hoop Nature Reserve. The lake has no surface outflow, and it is separated from the sea by 2.5 km of mobile sand (Lanz, 1997). The Sout River and a tributary, the Potberg River, drain into De Hoop Vlei from the north-west and east directions, respectively.

The increased availability of soils from agricultural activities has resulted in significant sediment deposition in the Sout River (The River Health Programme, 2011). The water quality of the upper segments of the Sout River is good and it deteriorates as the river flows downstream (The River Health Programme, 2011). This deterioration of downstream water quality may be due to the deposition of nutrient rich sediments from agricultural lands. The Sout River has been identified as one of the priority regions for initiatives aimed at conserving riparian vegetation in the Overberg (Herdien et al., 2005). This is because of its high conservation potential and low demands on rehabilitation time and costs (Herdien et al., 2005).

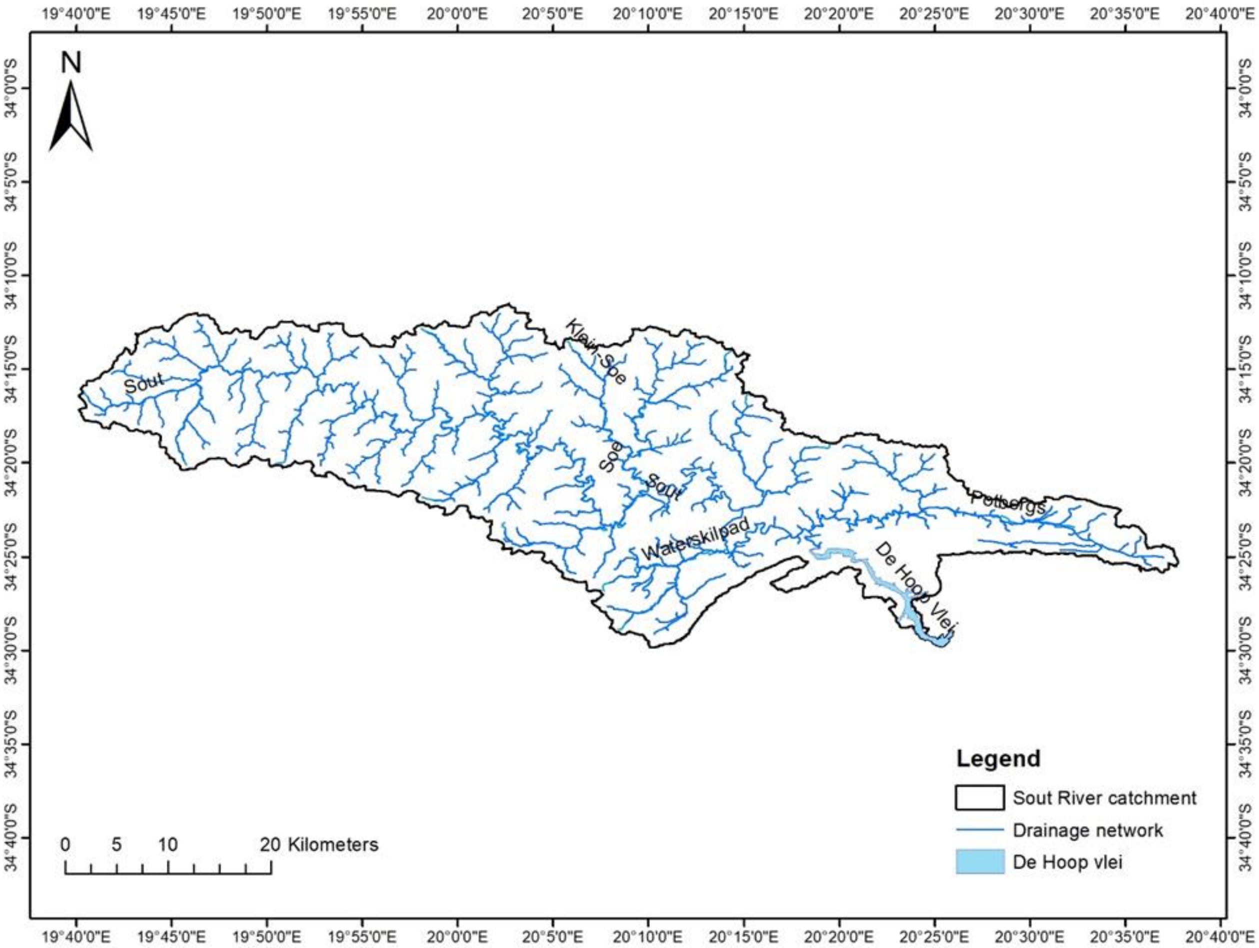


Figure 3.4: The drainage network showing minor and major tributaries of the Sout River catchment.

Chapter 4: Methods

4.1 Introduction

This chapter presents data collection and analysis methods that were applied in this study. The overall approach of this study is the combination of geospatial connectivity modelling, field sampling and lab analysis. The geospatial connectivity modelling was undertaken to fulfill the first and final objectives of the study, whereby the model was used to identify sediment pathways and ultimately high-risk contamination areas in the catchment. Field sampling and lab analysis were done to address the second, third and last objective which involve phosphorus concentrations and its longitudinal distribution in the catchment. Sediment samples were used to investigate the spatial variation of total adsorbed phosphorus.

4.2 Data collection methods

4.2.1 Geospatial data

Tiles of 5 m interval digital georeferenced contour data, and surveyed spot height data were sourced from National Geo-spatial Information (NGI), Mowbray, Cape Town. These data were interpolated to a 5 m resolution digital terrain model (DTM) using Topo to Raster in ArcMap 8.1, with no initial enforcement of drainage to fill sinks. The DTM was prepared for SedInConnect geospatial sediment connectivity modelling (Crema and Cavalli, 2018) using TauDEM Tools Version 5.3 (Tarboton et al., 2015). First, pit removal was implemented to ensure continuity of drainage required to compute flow directions and flow accumulation across the grid. Second, the Sout River catchment was extracted to a pour point located on the D8 flow accumulation maximum at the head of the De Hoop Vlei coastal lake, which is the sediment receiving environment of the Sout River catchment. De Hoop Vlei is impounded at the coast by barrier dune development, and there is no surface outflow from the lake to the Indian Ocean. The small area of catchment adjacent to the lake boundary was manually digitised using the NGI contours and merged with the catchment boundary polygon. Both the initial and pit filled DTMs were clipped on a 30 m buffer around the full catchment boundary to serve as inputs for SedInConnect modelling, as required by the model (Cavalli et al., 2013).

Finally, third-order drainage lines were extracted using a threshold flow accumulation value based on evaluation of drainage patterns using 0.5 m resolution digital colour orthorectified imagery, also from the NGI. This drainage line network served as the ‘target’ feature for subsequent connectivity modelling. Connectivity was therefore considered with respect to the probability of hillslope sediment entering the third and higher order drainage network, from where it could be transferred downstream. The Sout River catchment has numerous small farm dams that could play a role in trapping sediment before it reaches the drainage network. A polygon layer of farm dams was thus extracted from NGI vector hydrographic data, for use as ‘sinks’ input to the SedInConnect modelling. This removes areas draining into the farm dams from further analyses of connectivity. The main agricultural settlements of the catchment, and the boundary of De Hoop Vlei Lake, were manually digitised from the NGI orthorectified imagery for mapping purposes.

4.3.2 Vegetation cover data

IC calculates connectivity based on topographic characteristics such as slope and surface roughness. However, in lowland areas connectivity is driven by factors other than topography. Therefore, the index needs to be adjusted or used with other land use factors to better reflect connectivity in these areas. Since the focus of this study is to identify areas that should be prioritized for riparian buffer restoration, a method that incorporates IC and the vegetation cover in the catchment had to be adopted to evaluate the effects of vegetation on sediment connectivity to the drainage network. Additionally, the land use in the Sout River catchment is mainly agriculture, so using vegetation cover to adjust IC is more appropriate.

Normalized difference vegetation index (NDVI) was used to quantify vegetation richness in the catchment. The NDVI is a dimensionless index that quantifies the difference between red and near-infrared (NIR) light reflectance from vegetation. The index is used to assess the presence of live green vegetation cover. The cellular structure of the leaves highly reflects NIR while chlorophyll strongly absorbs visible light (Earth Observation System, 2022). This means that areas that are covered by live green vegetation will absorb more visible light while areas with bare ground will reflect more of the NIR. Therefore, observing the changes in NIR in comparison to red light provides a reliable indicator of vegetation cover on the ground (Earth Observation System, 2022). NDVI is calculated as follows:

$$\text{NDVI} = \frac{NIR-RED}{NIR+RED} \tag{1}$$

NIR is the reflection in the near-infrared spectrum.

RED is the reflection in the red range of the spectrum.

NDVI values range between -1 and 1, whereby the lowest values represent lack of green vegetation cover while the highest values represent dense green vegetation.

Four Sentinel-2 images captured on the 16th of December 2020 covering the Sout River catchment area were downloaded from the Copernicus website (Copernicus Open Access Hub, 2021). This date was chosen because these were the only recent (at the time of acquisition) images available on the website which were taken in the middle of the wet season when croplands are fallow covering the whole catchment area. The highest spatial band resolution, 10 m, was used in the analysis on the NDVI. The Sentinel-2 satellite can survey in the visible, near infrared, short-wave infrared spectral zones including thirteen spectral bands. This makes it possible to capture differences in vegetation state and minimizes the impact on the quality of atmospheric photography (Earth Observation System, 2022).

4.2.3 Sediment and water samples

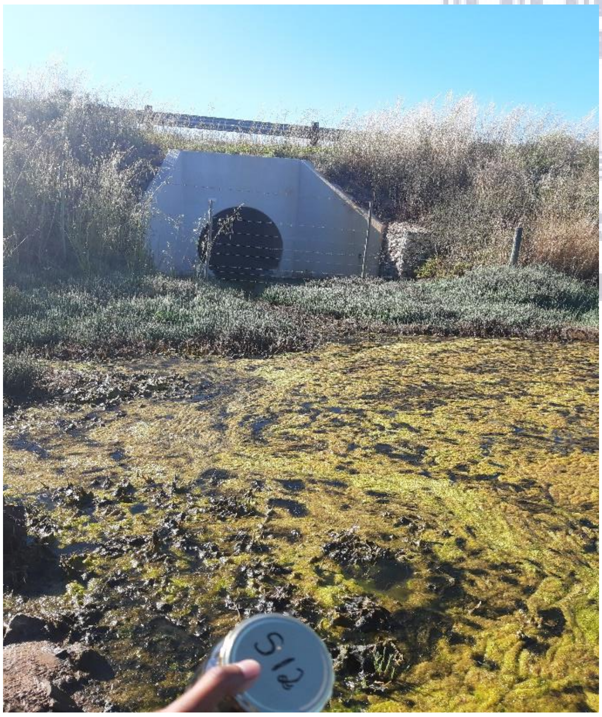
Field sampling of sediments and water samples in the Sout River catchment took place on the 18th and the 19th of October 2021. This is towards the end of the winter wet season and the beginning of the summer dry season in the area. The planting of wheat and canola (the main crops of the region) usually takes place at the beginning of the wet season, with the harvest taking place in late spring to summer (Department of Agriculture, Forestry, and Fisheries, 2016a; 2016b). River channel slack water sediment deposits sampled at the end of the wet season represent as far as possible the catchment response to antecedent (previous summer) vegetation cover characteristics (indicated by the NDVI) and the rainfall characteristics of the preceding wet season.

The images in Figure 4.1 were taken during the field trio in the Sout River catchment. Most parts of the river channel were covered with algae (Figure 4.1(a)and (c)). Figure 4.1(b) shows how most of the catchment looked like: large, cultivated areas with patches of the Renosterveld vegetation along the riparian margin. Figure 4.1(d) was taken in the Haarwegskloof Renosterveld Reserve which is the natural ‘reference’ site. The reserve was completely covered with Renosterveld.

a)

b)

c)



d)

Figure 4.1: The images were taken during sampling at the various sampling points. Picture a was taken at sampling point S15, b at S20, c at S12 and d was taken in the Haarwegskloof Renosterveld Reserve which is the natural ‘reference’ site.

Sediment samples were collected from slack water deposits on the channel bed to be analysed for adsorbed total phosphorus concentration, particle size distribution and organic carbon percentage. This was done longitudinally through the catchment and on tributaries where access permissions allowed but mostly a short distance upstream of road crossings. The first 24 (S1-S24) sampling sites (Figure 4.2) were chosen to cover as much of the catchment surface area as resources and access allowed. These sites include the main channel and tributaries of the river that run through agricultural land. The last two sampling sites (S25 and S26) are 'natural reference' sites located within the Haarwegskloof Renosterveld Reserve.

The Haarwegskloof Renosterveld Reserve is about 5 km^2 in size and some tributaries of the Sout River pass through the reserve. Since the tributaries are protected and are not directly impacted by agricultural activities that are taking place in the area, they serve as the only possible natural reference sites to measure background phosphorus concentrations in the catchment, for comparison with sites on agricultural land, as it is known that phosphorous concentrations in Renosterveld soils are naturally high (Thwaites and Cowling, 1987; Curtis-Scott et al., 2020). It is acknowledged that the available sampling sites within the reserve were on ephemeral tributaries that were dry at the time of sampling, and had high leaf-litter contents, and were therefore biophysically different from most of the sites sampled elsewhere in the catchment, which had at least some degree of flow or standing water. The reference sites thus provide the best available view of background phosphorous concentrations of upland hillslope soil transfers, which would naturally be subject to some degree of transformation during fluvial transport to lower reaches of the system. A full picture of natural background phosphorous concentrations reflecting variation in catchment position is not possible under the current land use conditions, and so the background concentrations presented are interpreted with some caution.

River surface water samples were collected simultaneously at the same sites except for the natural reference sites which were dry at the time of sampling – this was an unavoidable artefact of the nature of agricultural development in the region, replacing all but a very small percentage of mostly headwater and steeper slope Renosterveld environments (Curtis-Scott et al., 2020). The sediments were collected and stored in 250 ml glass jars while the water samples were stored in airtight plastic bottles in a chest freezer prior to analysis. A Garmin etrex 22x Global Positioning System (GPS) was used to capture the location of each sampling site with 3 to 5 m accuracy.

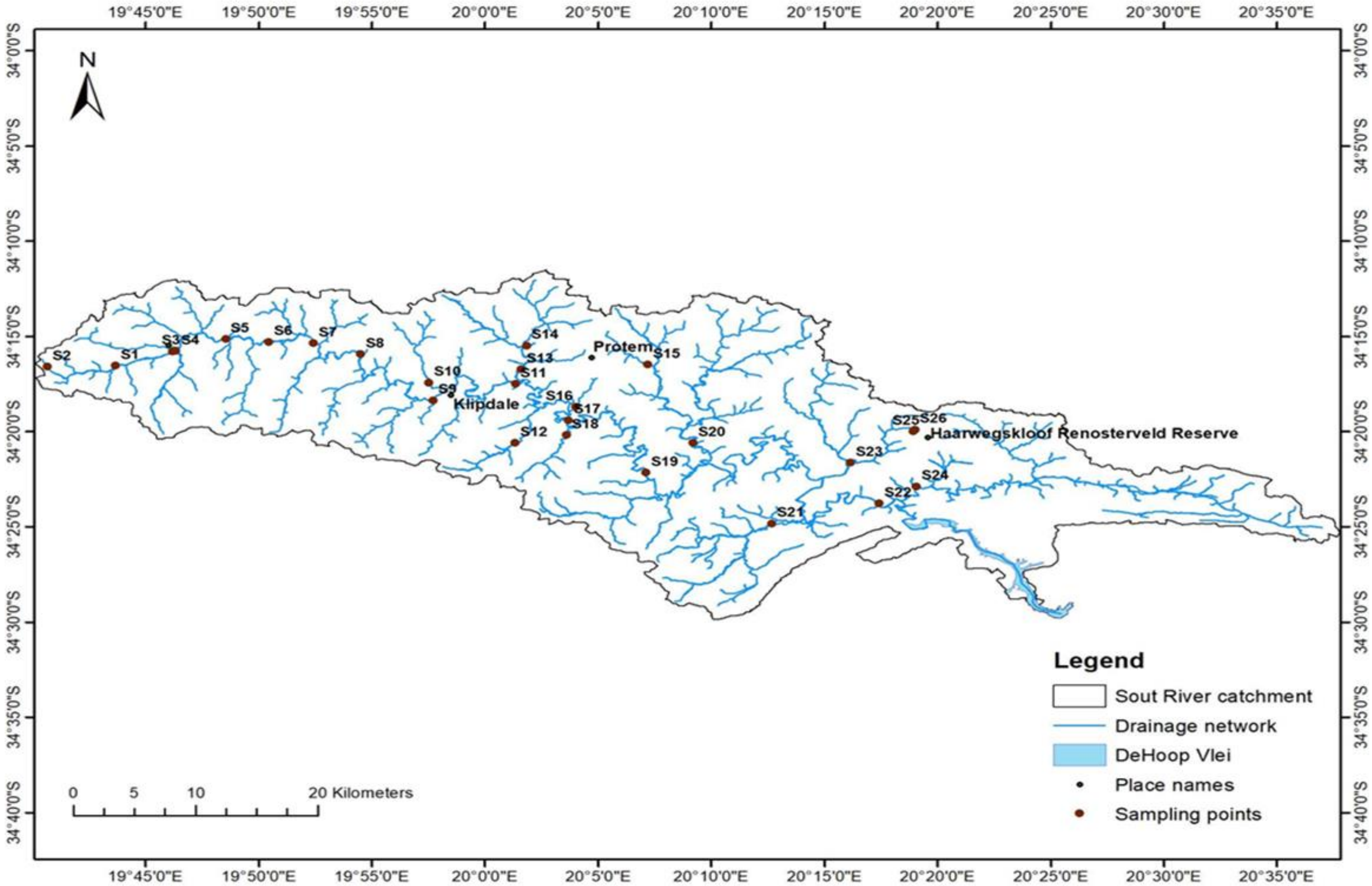


Figure 4.2: The locations of the sediment and water sampling points in the Sout River catchment

4.3 Data Analysis

4.3.1 Sediment connectivity analysis

To determine the potential sediment connectivity in the catchment, the modified version of Borselli et al. (2008) index of connectivity was used. Figure 4.3 below outlines the workflow that was followed during the sediment connectivity analysis. Additionally, the workflow diagram also outlines the NDVI and IC integration. IC was computed using a geospatial terrain analysis tool called SedInConnect. SedInconnect is a freeware software that was developed by Crema and Cavalli (2018). It determines the IC of Borselli et al. (2008) by implementing the modifications proposed by Cavalli et al. (2013) with further refinements.

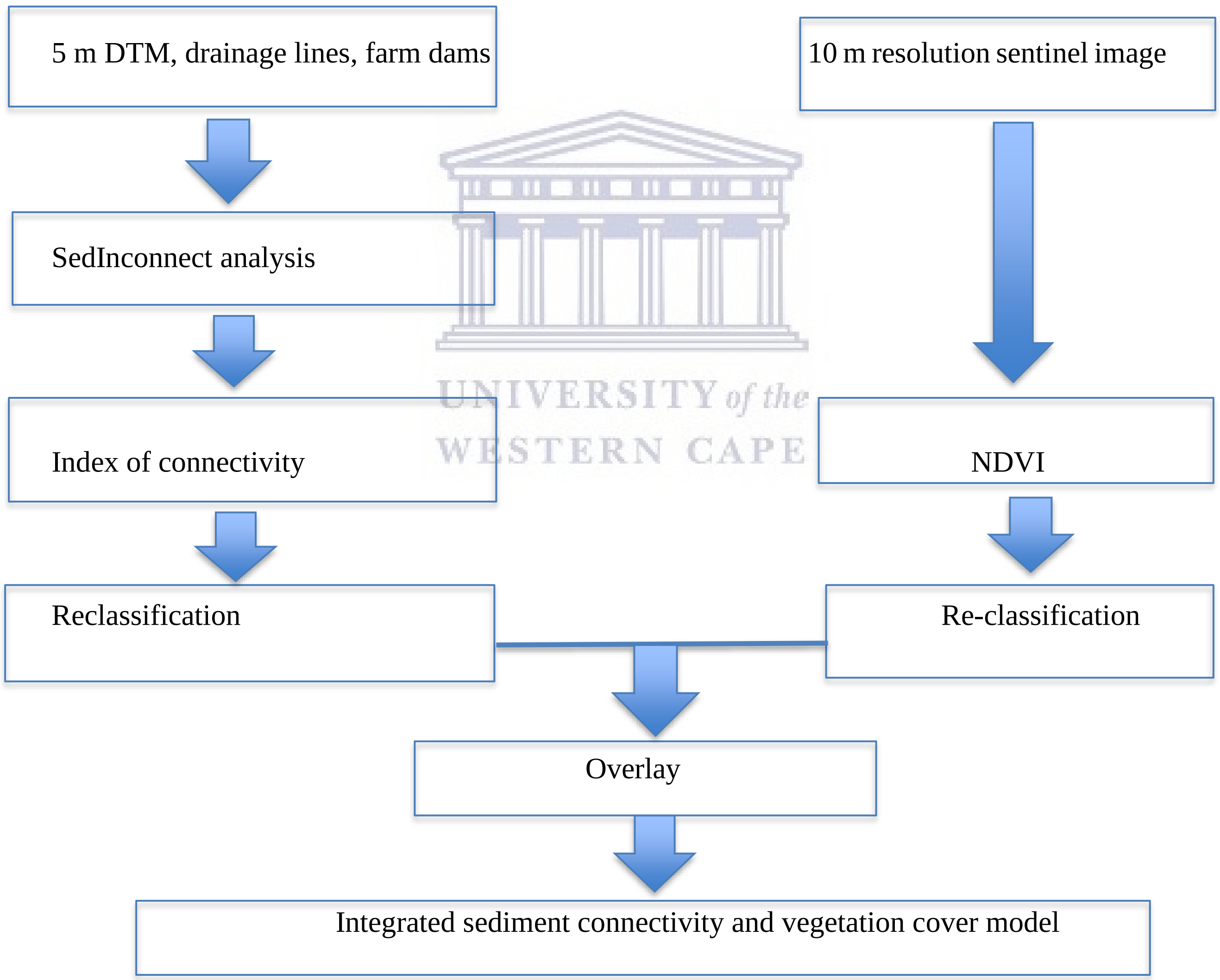


Figure 4.3: A summary of the workflow followed when developing the integrated sediment connectivity and vegetation cover model.

The tool is based on the relationship between the upslope and downslope components of sediment connectivity within a catchment. The upslope component (D_{up}) quantifies the potential for the downward movement of sediments that are eroded upstream due to upslope catchment area A, mean slope S, and an impedance factor W (Eq. 3). The downslope component (D_{dn}) is quantified as the sum of the ratio of downslope flow along the ith cell according to the steepest downslope direction from the distance a particle must travel to arrive at the nearest target (Eq. 4). This method quantifies connectivity as the likelihood of sediment being mobilized from a certain point in a catchment to a specific target. IC is quantified as the log10 ratio of the upslope to downslope components of connectivity (Eq. 2).

$$IC = log_{10}(\frac{D_{up}}{D_{dn}}) \quad (2)$$

$$D_{up} = \underline{WS}\sqrt{A} \quad (3)$$

$$D_{dn} = \sum_i \frac{d_i}{W_i S_i} \quad (4)$$

Where, $\bar{W}$ is the average weighting factor, which represents the impedance of runoff and sediment fluxes due to surface roughness. $\bar{S}$ is the average slope gradient and A is the upslope contributing area. D_i is the length of the flow path along the ith cell according to the steepest downslope direction. W_i is the weighting factor while S_i is slope gradient of the ith cell. The values of IC can range between -∞ and +∞ where sediment connectivity increases with higher IC values.

The tool uses a structural connectivity assessment, presuming that connectivity is a function of the continuity—or lack thereof—of runoff and sediment paths at a particular point in time. Although the tool assumes that the main variable regulating runoff and sediment paths serving as an impedance factor, W, is topography related, it can also be run with user-selected, process-related impedance factors (Crema and Cavalli, 2018). This makes it an attractive tool to use in difference environmental settings.

Topography-based IC computation

Cavalli et al. (2013) introduced two scenarios for the application of the modified index, and they are ‘IC outlet’ and ‘IC channel’. ‘IC outlet’ refers to the analysis of sediment connectivity across the whole catchment between hillslopes and catchment outlet (IC outlet) while ‘IC channel’ refers to the analysis of sediment connectivity between hillslopes and the channel network. The present study analyzes sediment connectivity between hillslopes and the channel network, i.e., ‘IC channel’, to determine the probability of sediment transfer from hillslopes to the drainage network an analysis of target connectivity was performed. The index of connectivity was chosen for this analysis for its minimal data requirements and straightforward application.

Figure 4.4 below shows the SedInConnect 2.3 user interface. The 5 m pit filled DTM was used as an input layer into the SedInConnect software. The original 5 m DTM was used to compute a topographic roughness-based weighting factor following the approach by Cavalli et al. (2013). A polygon layer consisting of farm dams in the Sout River catchment was inserted as a ‘sinks’ layer, while the drainage network was inserted as the ‘target.’ Normalization of the weighting factor followed the method proposed by Trevisani and Cavalli (2016).

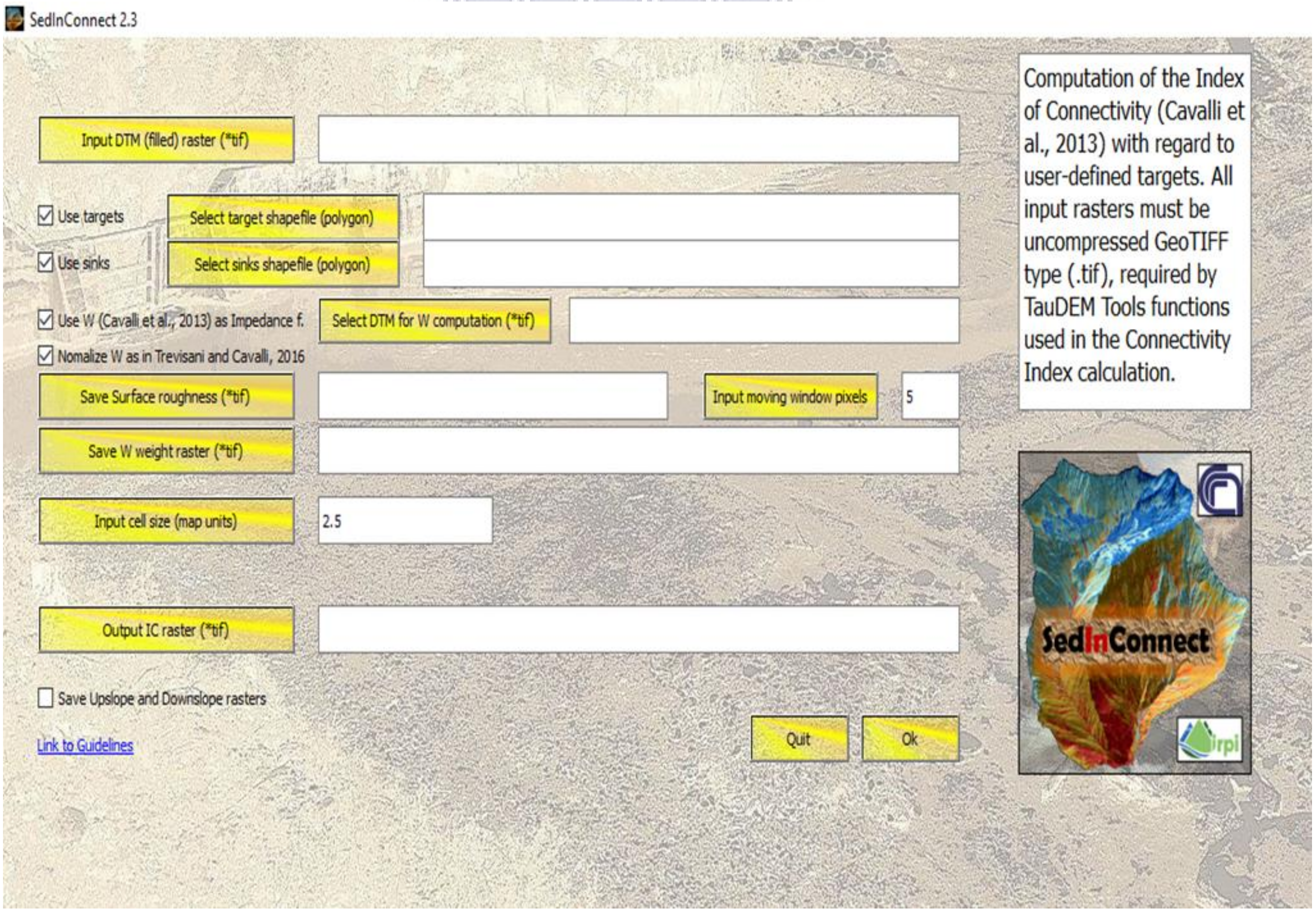


Figure 4.4: SedInConnect version 2.3 user interface showing the required input data.

Figure 4.5 below displays the parameters that were input into the model to compute IC; 5m DTM, a polygon layer of farm dams (sinks) and the drainage network of the Sout River catchment (target). The output IC map was classified into four categories of low, medium low, medium high and high using the Jenks Natural Breaks classification (Tiranti et al., 2018) in ArcMap 8.1.

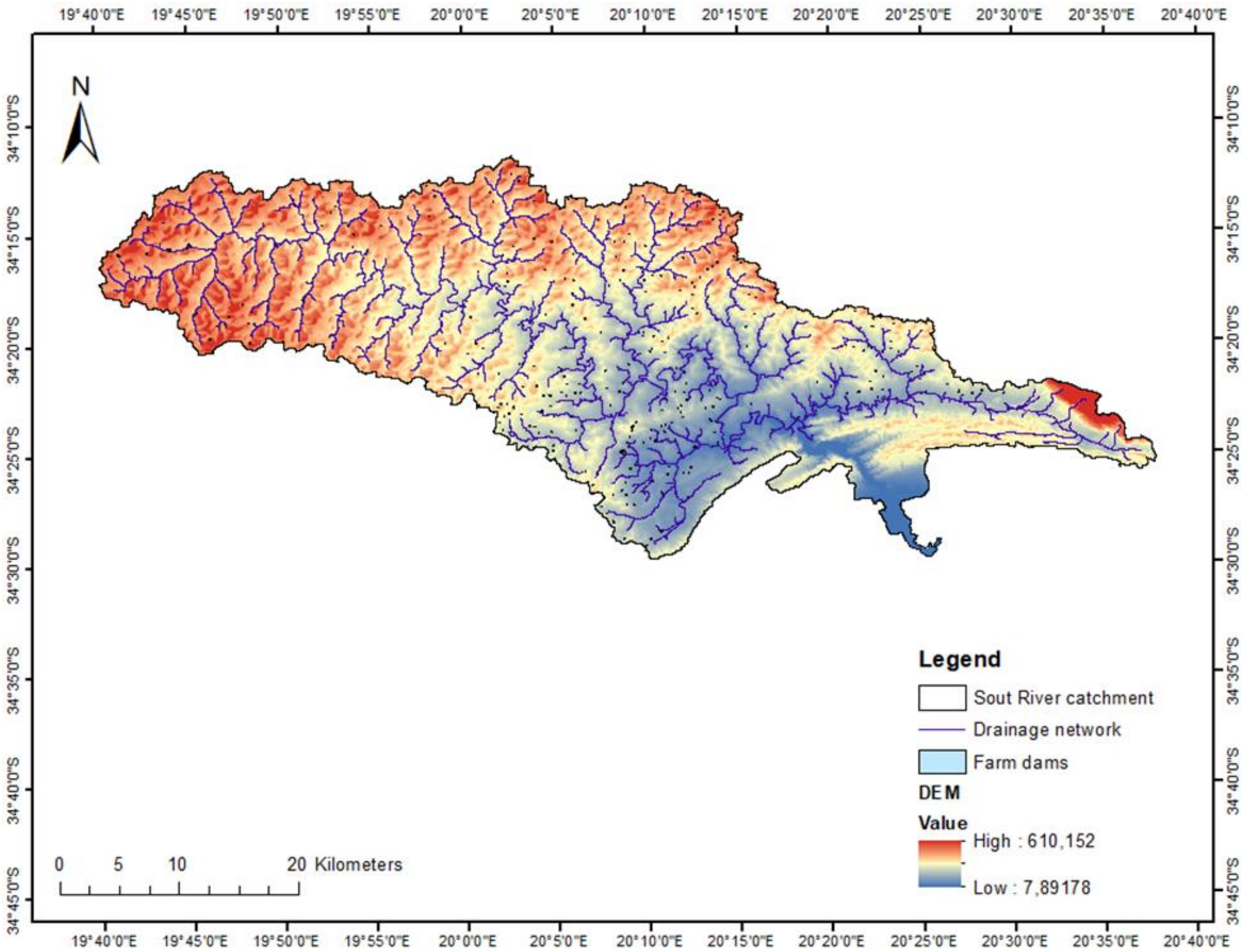


Figure 4.5: The original DEM, farm dams and drainage network used in the computation of IC in SedInConnect.

NDVI

The raster calculator toolbox in ArcMap 8.1 was used to calculate the NDVI of each of the 10 m resolution Sentinel-2 images in accordance with Eq. 1. The resulting NDVI images were then mosaicked to create one image which was then clipped to the catchment boundary to produce an image that covers the whole catchment area. The image was classified according to the NDVI values to represent land (vegetation) cover. All these analyses were done using ArcMap 8.1.

Integrated sediment connectivity and vegetation cover model

To compute the sediment connectivity and vegetation cover model an overlay analysis was performed using ArcMap 8.1. To begin the process, the pixels of the topography computed IC and the NDVI maps were reclassified using the spatial analysis tool. The IC classes that were labeled as 'Low,' 'moderately low,' 'moderately high' and 'high' were assigned new values which were 0, 1, 2 and 3, respectively. The classes were assigned new values based on their level of sediment connectivity, i.e., the higher the IC, the higher new class value and vice versa. The barren land, shrub and grassland, as well as dense vegetation classes of the NDVI map were assigned new values 2, 1 and 0, respectively. Just like the IC map reclassification, the new values were assigned for each class based on how well sediments can move through the land(vegetation) cover and reach the river channel. Therefore, higher values were assigned to classes with the least vegetation cover, while the lowest values were assigned to classes with the highest vegetation cover. This was to indicate that vegetation cover reduces sediment transport. Water bodies on the NDVI map were removed from the analysis because they are considered sediment sinks.

Once the reclassification process was complete, the new IC and NDVI maps were added together in ArcMap 8.1 using the Map Algebra tool. This analysis resulted in a map with six classes with values ranging from 0 to 5 indicating the risk of sediment contamination due to potential sediment pathways and vegetation cover in the Sout River catchment. The classes were then labelled 'very low', 'low', 'medium-low', 'medium-high', 'high' and 'very high' based on the risk of sediment contamination.

4.3.2 Analysis of sediments for total adsorbed phosphorus concentration

The sediment samples were oven-dried at 105°C for 24 hours. Samples were crushed with a pestle and mortar, homogenized and sub-sampled. One subsample was sieved through a 63µm sieve to isolate the sediment size fraction to which phosphorus tends to preferentially adsorb (silt and clay). This material was analyzed for adsorbed total phosphorus concentration using inductively coupled plasma atomic absorption spectrometry (ICP-AES). This method analyses samples in liquid form. Therefore, microwave acid digestion was used to extract adsorbed phosphorus for analysis. The equipment used was a Thermo Icap 6300 ICP-AES which analyses concentrations of major and minor elements.

4.3.3 Analysis of water samples for orthophosphate concentration

Orthophosphate concentration in the water samples was analyzed using the Molybdovanadate method which was adapted from Standard Methods for the Examination of Water and Wastewater (Hach, 2019). This method works by reacting orthophosphate with molybdate in an acid medium to produce a mixed phosphate/molybdate complex (Hach, 2019). Yellow molybdovanadophosphoric acid is generated when vanadium is present. The yellow colour intensity is proportional to the phosphate concentration. This analysis was done using a Hach spectrophotometer with a wavelength of 430 nm (Hach, 2019).

To prepare the blank 10 ml of ultrapure water and 0.5 ml of Molybdovanadate reagent were added to a sample cell. The mixture was then swirled to allow the chemicals to mix and was left to react for 7 minutes. After the 7 minutes had lapsed the sample cell was cleaned with a paper towel and thereafter inserted into the instrument's cell holder. The zero button was pushed, and the display showed 0.0 mg/l PO4. This was done to establish a reference concentration against which other samples were evaluated. The same procedure was followed when preparing the samples except the water samples were used instead of deionized water. The 'zero' button was also not pushed in this case, instead the 'read' button was pushed, and the results were displayed in mg/l.

4.3.4 Measurement of sediment organic carbon by loss on ignition (LOI)

The loss on ignition is a common method that is used to provide an estimation of the organic content in soils and sediments (Heiri et al., 2001). In this method, sediment samples are combusted at a high temperature (500-550 ℃) making the organic matter content oxidise into carbon dioxide and ash (Heiri et al., 2001). The weight loss during the reaction is then measured to provide an estimate of the organic matter content in the sediment. Factors such as sample size, furnace temperature and ignition duration all affect the results of the LOI. Nonetheless, the method was found to be a quick and inexpensive technique for estimating organic matter content (Heiri et al., 2001).

In the present study, loss on ignition measurement was based on weight losses after combustion at 550 ℃. The sediment samples were oven dried at 105 ℃ for 24 hours. Dry and empty crucibles were weighed and labelled according to the sample labels (S1-S26). This weight was then recorded as W1. A spoonful of each dry sediment sample was put into each crucible and weighed. This weight was recorded as W2. The crucibles were put on a tray and transferred into a blast furnace to run at 550 °C for 6 hours. After 6 hours the furnace was turned off and the samples were left in the furnace to cool overnight. Once the samples had cooled, they were weighed again, and this weight was recorded as W3.

The Loss on Ignition percentage was calculated for each sample as follows:

$$\%LOI = \frac{W_2 - W_3}{W_2 - W_1} \times 100 \qquad (5)$$

Where LOI represents loss in ignition percentage at 550 °C, W_1 represents the dry weight of the sample before ignition, W_2 represents the dry weight of the sample after ignition.

Figure 4.6: *S*howing the samples in the oven and one sample on the mass balance during the LOI analysis.

4.3.5 Analysis of sediment particle size

Another subsample of homogenised crushed sediment was sieved through a 1 mm sieve. Two spoonfuls of the sieved sediments were transferred into a 500 ml beaker labelled with the relevant sample code. 30 ml of hydrogen peroxide was added to each beaker and carefully swirled to ensure that the sample made contact with the peroxide. The beakers were placed into a fume hood. This process was done to remove any organic matter from the samples. Organic matter in the sample reacted with the peroxide causing frothing and deionized water was added into the beaker when the frothing threatened to overtop the beaker. The beakers were then removed from the fume hood and 5 ml of hydrochloric acid was added to each beaker to disperse metallic binding agents. Once the samples had cooled down, they were filtered through a 1 µm filter paper to separate the sample from the waste. The filtered sediment samples were then transferred into centrifuge tubes that were labelled with the sample codes. Particle size was analysed using a Micrometrics Saturn DigiSizer which has a liquid input system and inbuilt sonicator and uses laser diffraction methods. The particle sizes were classified into clay, silt, and sand.

4.3.6 Statistical analysis

The relationships between sediment adsorbed phosphorus concentration and water orthophosphate concentration, organic carbon percentage and sediment particle size were investigated using Spearman's Rank Correlation tests. The tests were done using the Statistical Package for Social Science (SPSS v28) software at 95% confidence level. The Spearman's Rank Correlation test was chosen because it is a nonparametric test that is used when the data of some of the variables are not normally distributed, which was the case in this study. The correlation tests were done to determine whether these variables influence the amount of phosphorus that adsorbs to the sediments or not. This knowledge is necessary for phosphorus management in the catchment.

4.3.7 Comparison of the concentrations of adsorbed phosphorus and background levels at a natural 'reference' site

The concentrations of adsorbed phosphorus at the sampling sites (S1 to S24) were compared with those at the 'reference natural' sites (S25 and S26) located within the Haarwegskloof Renosterveld Reserve. This comparison was achieved by calculating the difference of the average background adsorbed phosphorus (494 mg/kg) concentration and the concentrations measured at the different sampling sites. The difference of those values was then mapped and classified based on whether they were higher or lower than the average background phosphorus concentrations.

4.3.8 Identification of high-risk areas of adsorbed phosphorus contamination in the Sout River catchment.

The identification of high-risk areas of phosphorus contamination in the catchment was informed by the combination of catchment surface characteristics (sediment connectivity and vegetation cover) and sediment-bound phosphorus concentration. To identify high risk areas for adsorbed phosphorus contamination in the catchment, the integrated sediment connectivity and vegetation cover map was overlain with the difference in adsorbed phosphorus concentration map in ArcMap 8.1. In this case, areas that are highly connected and have high concentrations of adsorbed phosphorus were considered high risk and areas with low connectivity and low adsorbed phosphorus concentrations were considered low risk.

Chapter 5: Results

5.1 Introduction

This chapter presents results which address the four objectives of this study in the order in which they are presented in Chapter 1. The first objective is addressed by presenting maps that display topography-computed IC and another map displaying the integration of IC and vegetation cover. The second objective is addressed by presenting a map that shows the spatial variation of adsorbed phosphorus in the catchment and further investigates factors that may be influencing these concentrations. A map that displays the differences of background phosphorus and phosphorus concentrations at the sampling sites is used to address the third objective. Finally, to address the last objective, the risk of contamination map is overlain with the adsorbed phosphorus map. This was done to identify high risk areas in the catchment that have high IC and high phosphorus concentrations, and therefore, have high risk of contamination, and should thus be prioritized for riparian buffer restoration.

5.2 The spatial variation in sediment connectivity in the Sout River catchment

Figure 5.1 presents the index of connectivity map for which the pixel values are ranked into four classes according to quartile boundaries using the Jenks Natural Breaks Classification in ArcMap version 10.8.1. The IC values range from -6.71 to 3.53 with the mean and standard deviation being -4.21 and 0.80, respectively. According to the connectivity map, areas with the highest IC values occur along the drainage network and the connectivity decreases with distance from the drainage network. The southern side of the catchment has lower overall target connectivity compared to the northern side of the catchment where the highest IC values are distributed. Furthermore, the upper part of the catchment also displays lower overall connectivity values.

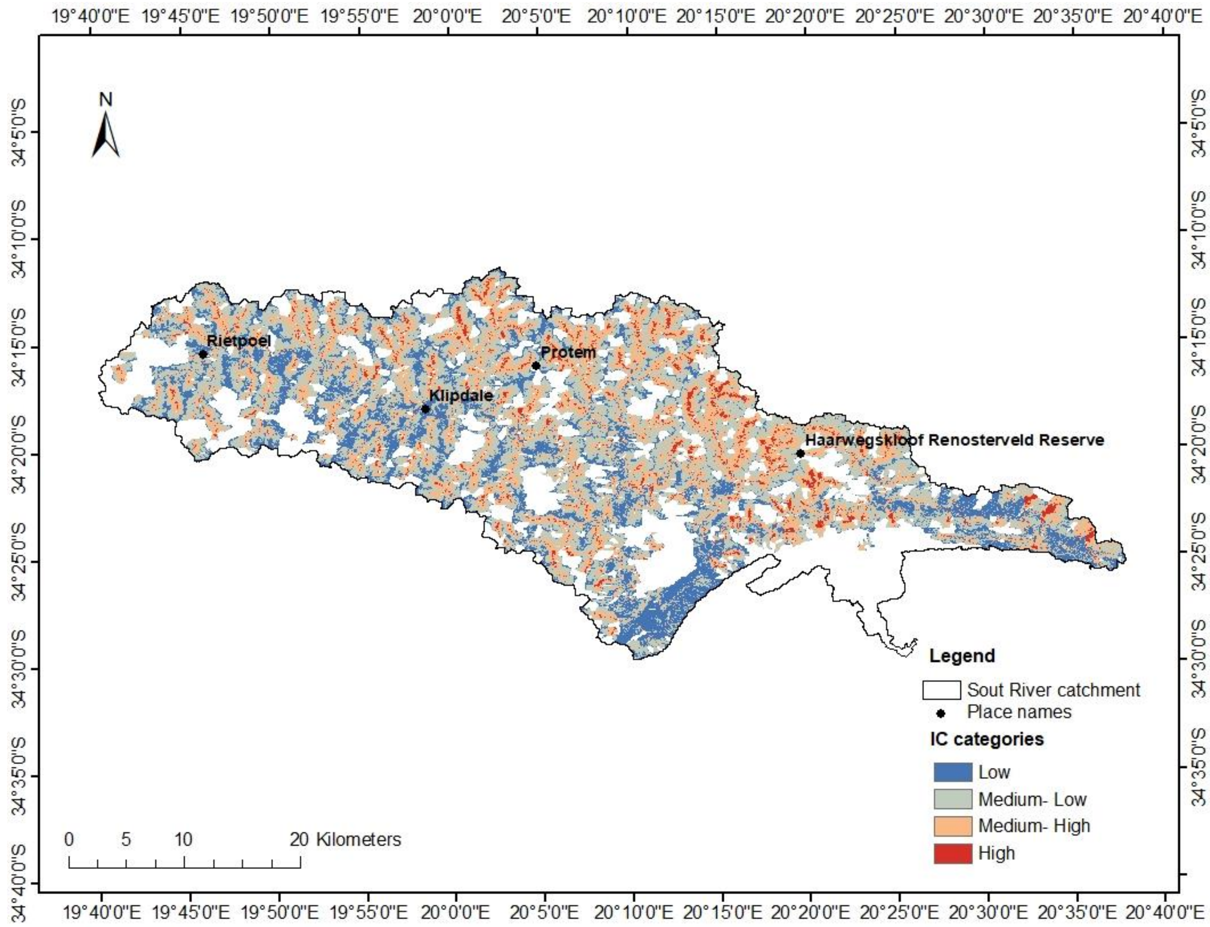


Figure 5.1: The spatial distribution of index of connectivity in the Sout River catchment. The map was computed using SedInconnect software, with the IC categories classified as 'Low,' 'Medium-Low,' 'Medium-High' and 'High'.

Table 5.1 summarizes the IC category ranges in terms of their pixel percentages on the IC map. Most of the IC values in this catchment are negative. This indicates that the potential of sediment transfer in the catchment is generally low. From the table, the 'Low', 'Medium-Low', 'Medium-High' and 'High' IC categories occur in the ranges -6.71 to -4.74, -4.74 to -3.9, -3.9 to -2.81 and 2.81 to 3.53, respectively. The 'High' IC category contains the least pixel percentage (4.54%) on the map. This means that there are very few areas in the catchment with hillslopes that are highly connected to the channel. This is followed by the 'Low' IC category whose pixel percentage is 22.91%. The 'Medium-High' IC category has the second most pixels with a percentage of 27.23% while the 'Medium-Low' IC category has the most pixel percentage (45.33%). Overall, the IC in this catchment can be regarded as being medium-low.

Table 5.1: A summary of the IC category ranges and their percentages.

IC category	Range	Pixel count	Pixel percentage (%)
Low	-6.71 - -4.74	8411259	22.91
Medium-Low	-4.74 - -3.9	16643376	45.33
Medium-High	-3.9 - -2.81	9997651	27.23
High	-2.81 - 3.53	1666192	4.54

NDVI

NDVI was used to model vegetation cover in the Sout River catchment. The resulting map is presented in Figure 5.2. The NDVI classes range between -0.5 to 0, 0 to 0.18, 0.18 to 0.6 and 0.6 to 0.91 for water body, barren land, shrub and sparse vegetation, and dense vegetation, respectively. Dense vegetation mainly occurs on the lower part of the catchment in De Hoop Nature Reserve and on the area with a higher relief which is situated on the north-eastern side of the catchment. A few patches of dense vegetation can also be observed on the southern side of the catchment. Overall, the catchment is dominated by barren land and shrub and sparse vegetation.

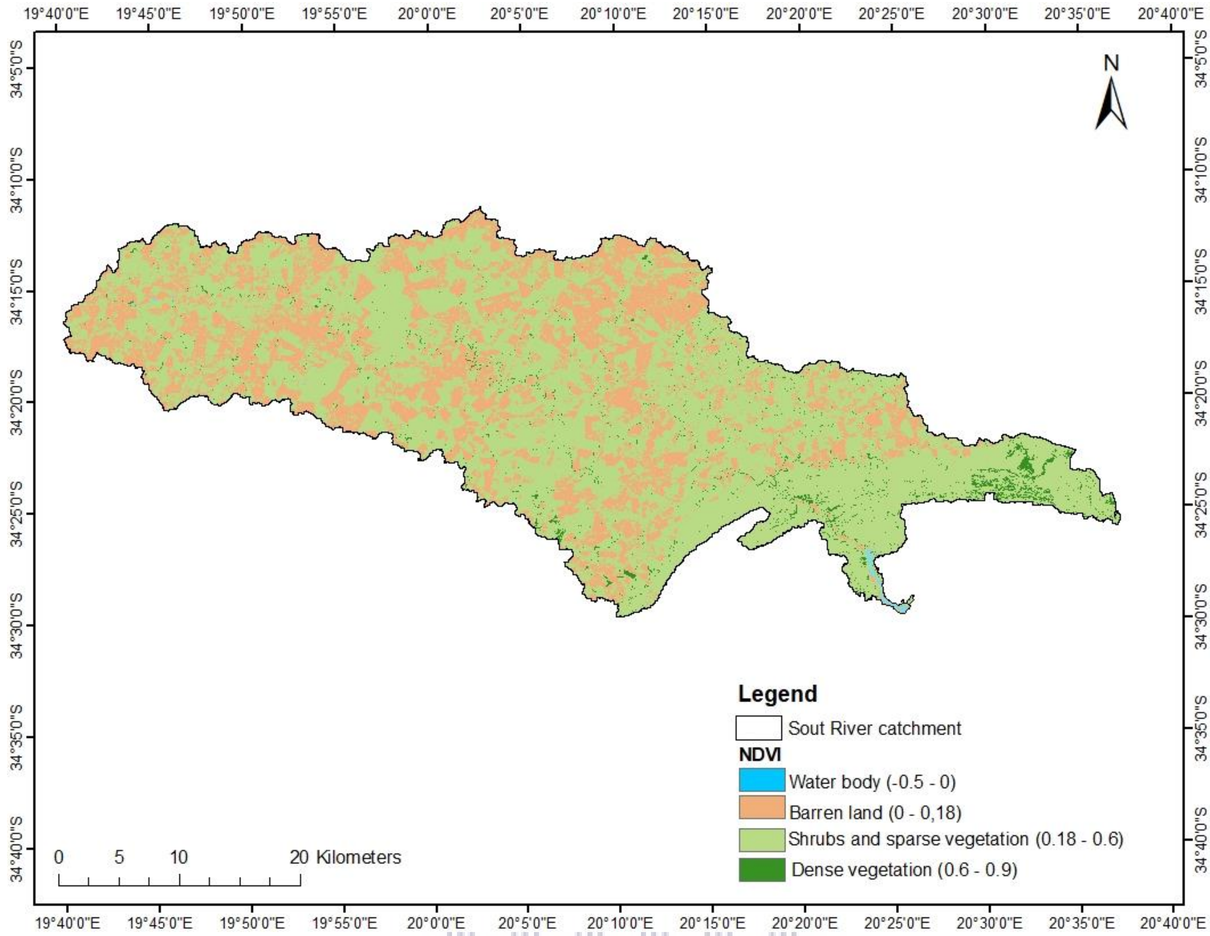


Figure 5.2: The NDVI of the Sout River catchment during summer (16 December 2020).

Integrated sediment connectivity and vegetation cover model

The result of the integration analysis of the topography-based IC and the NDVI is shown in Figure 5.3. The resulting map (referred to risk of sediment contamination) has a mean of 2.3 and standard deviation of 1.03. The map is classified into areas with 'Very low', 'Low', 'Medium low', 'Medium high', 'High' and 'Very high' sediment contamination risk based on potential sediment pathways (IC) and vegetation cover (NDVI). The classes account for pixel percentages 4.14%, 17.23%, 35.49%, 31,88%, 10.45% and 0.8%, respectively (Table 5.2). This indicates that the catchment has an overall medium risk of sediment contamination as far as sediment connectivity and vegetation cover are concerned. However, there are quite several areas on the northern side of the catchment that have a high risk of sediment contamination and these areas are located around Protem. Most areas with lowest risk of sediment contamination are located on the south and southeastern parts of the catchment. Klipdale and a few surrounding areas also have low sediment contamination risk. The Haarwegskloof

Renosterveld Reserve and surrounding areas ('natural reference site') show a medium-low contamination risk.

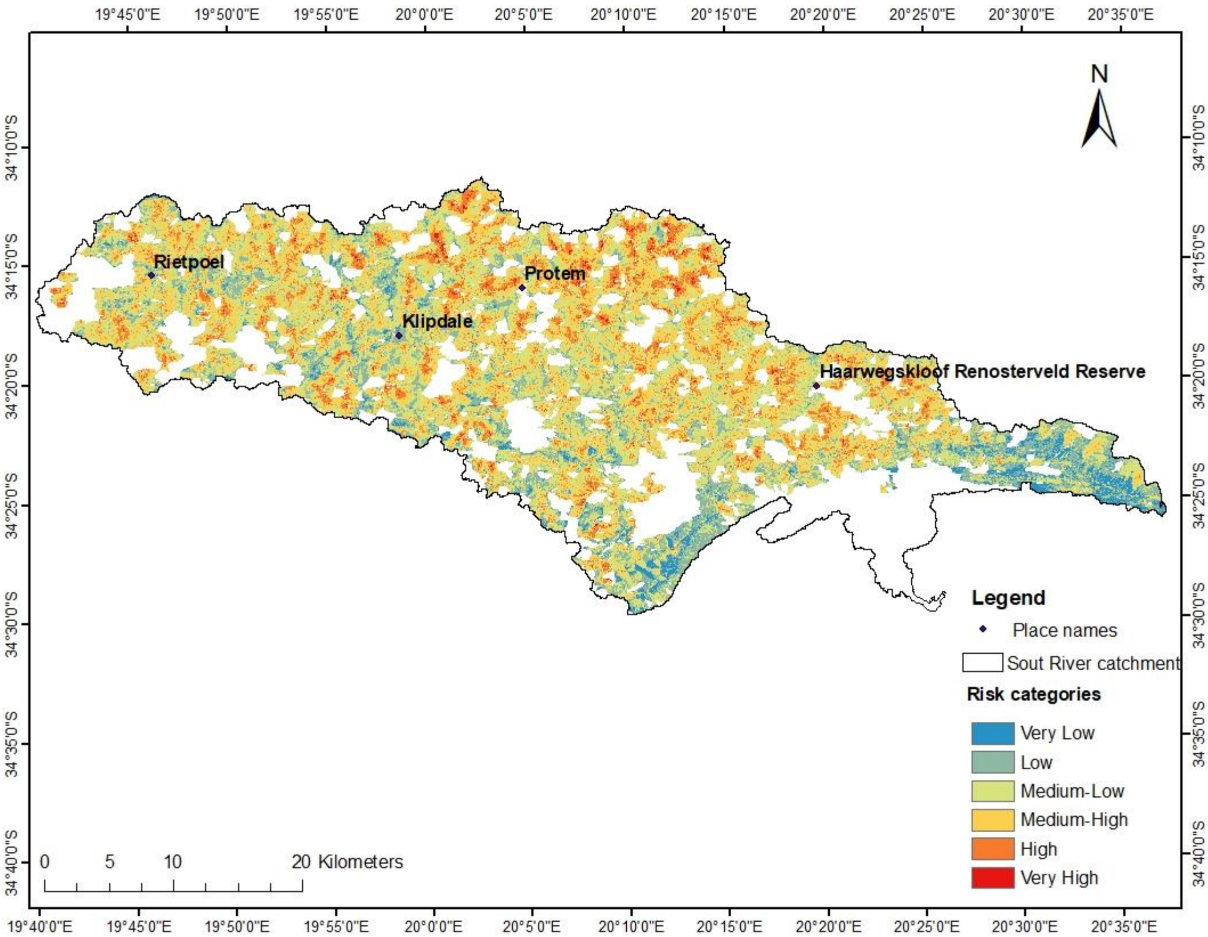


Figure 5.3: The risk of sediment contamination model computed using the integration of sediment connectivity (IC) and vegetation cover (NDVI).

Table 5.2: Pixel percentages of the contamination risk map categories.

Contamination risk	Pixel count	Percentage (%)
Lowest	395411	4.14
Low	1644867	17.23
Medium-Low	3388262	35.5
Medium-High	3043167	31.88

High	997654	10.45
Highest	76450	0.8

5.3 The longitudinal variation of adsorbed phosphorus

Adsorbed phosphorus concentrations varied from 237 mg/kg to 829 mg/kg with the median of 520 mg/kg. The spatial variation of adsorbed phosphorus concentration in the catchment is presented by the map in Figure 5.4. The circles on the map represent adsorbed Phosphorus concentration classes (see the map legend). The bigger the circle, the higher the concentration and vice versa. The lowest concentrations of phosphorus were observed upstream near the river source while higher concentrations were found to be in the middle and the lower reaches of the catchment. In other words, the concentration increases with distance from the river source. The highest concentration was found in sampling site S15 which is located on the Witklip se Loop tributary while the lowest concentration was measured at S6 which is located on the upper part of the catchment (Table 5.4 and Figure 5.7).

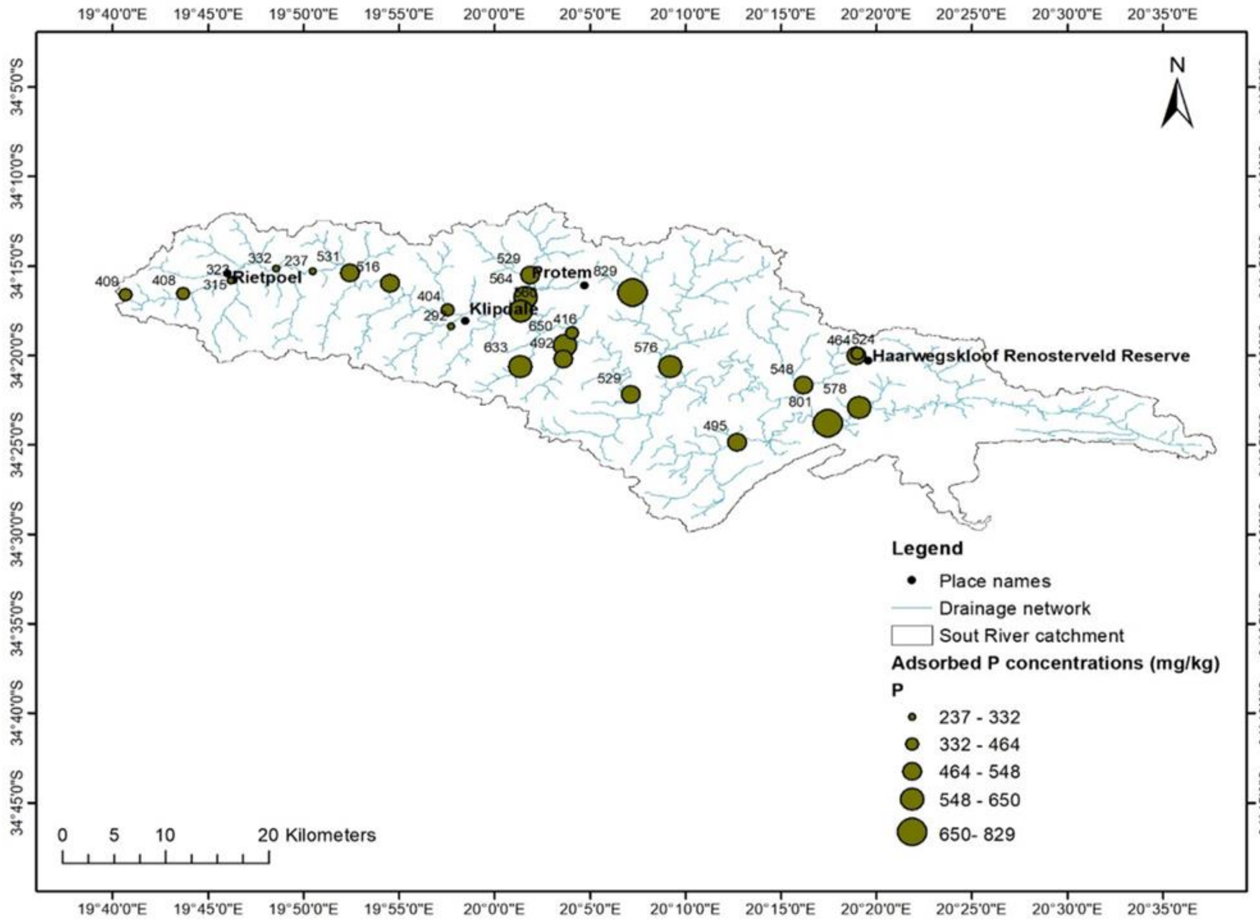


Figure 5.4: Longitudinal variation of adsorbed phosphorus concentrations in the Sout River catchment.

5.2.1 Potential factors that might influence the concentration of phosphorus in the sediments

There are various sediment composition factors that contribute to the amount of phosphorus that adsorbs to sediments, and in turn, influence its spatial distribution in the catchment. These factors were correlated with adsorbed phosphorus concentrations using Spearman's rank correlation to determine the existence of any relationship between the variables. The results of the correlations are presented in table 5.3 below. Scatter plots are also presented in Figure 5.5 to demonstrate the relationship between adsorbed phosphorus and the various variables.

Phosphorus occurs in both organic and inorganic forms, with organic forms accounting for a huge portion of the nutrients accumulated in the sediments (Zhu et al., 2013). Furthermore, phosphorus sorption at the sediment-water interface has been identified as an important process that influences nutrient concentrations in sediments (Yang et al., 201). Although researchers have reported contradicting results, organic matter content is another sediment composition factor that is associated with adsorbed phosphorus concentrations. In this study, organic matter composition was measured using the loss on ignition method and the percentages were found to range between 2% and 37.74% with the mean of 7.68% (Table 2 in the Appendix). The results obtained were correlated with adsorbed phosphorus concentration to investigate their association. The Spearman's Rank correlation test revealed that there was no statistically significant correlation between organic matter and adsorbed phosphorus concentrations in the Sout River catchment (r_s = 0.25, P=0.3).

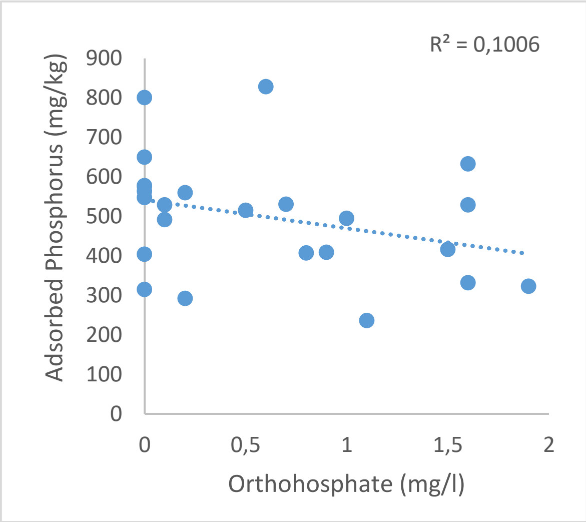
Table 5.3: Spearman Rank correlation test between adsorbed phosphorus concentrations and the various variables

Variable	r_s	p-value
Organic matter content	0.25	0.3
Clay	-0.02	0.91
Silt	-0.29	0.16
Silt and Clay	-0.19	0.34
PO_4	-0.37	0.08
pH	0.03	0.91
EC	0.21	0.38
Al and Fe	0.5	0.79

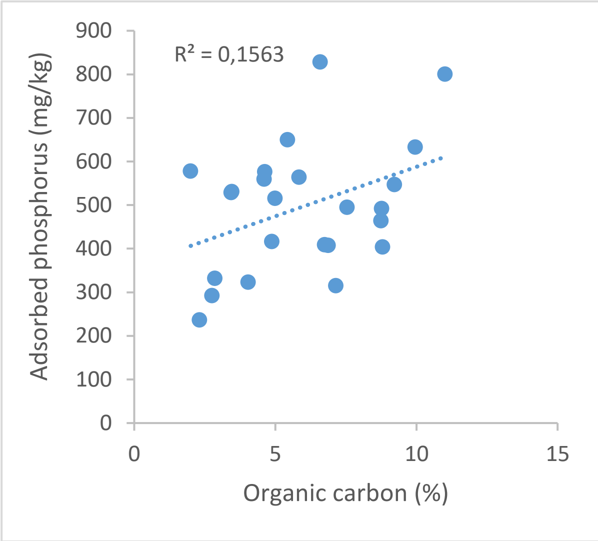
Smaller particle sizes such as clay and silt have been previously reported to be strongly associated with high concentrations of phosphorus due to their large specific area. Therefore, to study the relationship between particle size and adsorbed phosphorus concentrations clay, silt and the sum of clay and silt were correlated with adsorbed phosphorus concentrations using the Spearman's rank correlation test. The results revealed that there was no significant correlation between the variables (clay: r_s = -0.02, P=0.91; silt: r_s = -0.29, P=0.16; silt and clay: r_s = -0.19, P=0.34). This indicates that small particle sizes are not associated with phosphorus concentrations in this catchment and therefore more factors need to be explored.

Additionally, adsorbed phosphorus concentrations in the sediments were correlated with orthophosphate concentration in the river water samples. This was done to investigate if there was any association between the concentrations of phosphorus found in the sediments and in the water. This information is important as it can be used to determine the impact of phosphorus-bound sediments in the river. The correlation test revealed that there were no statistically significant relationships between adsorbed phosphorus concentrations and orthophosphate (r_s = -0.37, P=0.08) in this catchment.

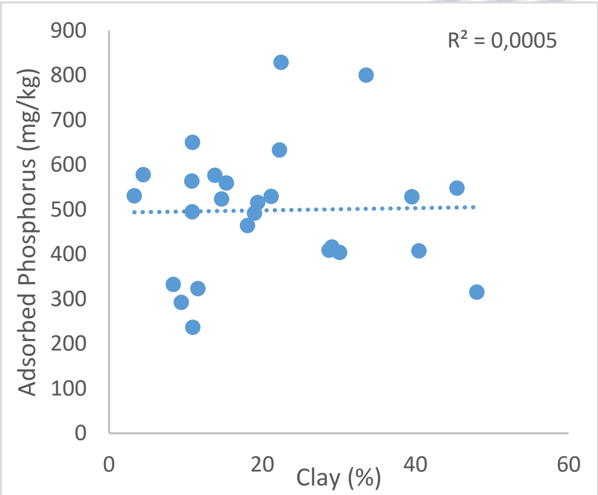
a)


R² = 0,1006
Adsorbed Phosphorus (mg/kg)
900
800
700
600
500
400
300
200
100
0
0
0,5
1
1,5
2
Orthohosphate (mg/l)

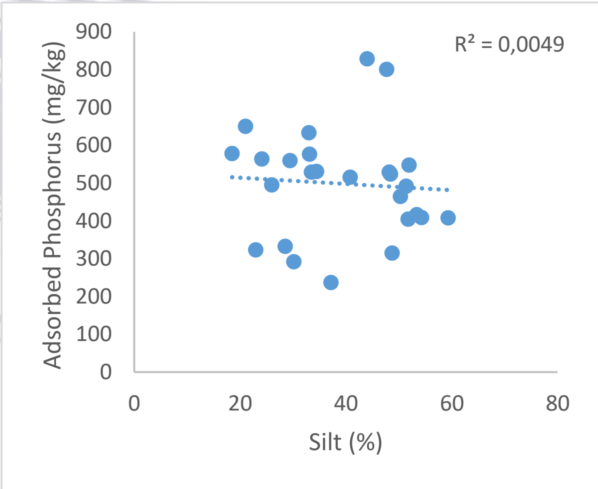
b)


R² = 0,1563
Adsorbed phosphorus (mg/kg)
900
800
700
600
500
400
300
200
100
0
0
5
10
15
Organic carbon (%)

c)


R² = 0,0005
Adsorbed Phosphorus (mg/kg)
900
800
700
600
500
400
300
200
100
0
0
20
40
60
Clay (%)

d)


R² = 0,0049
Adsorbed Phosphorus (mg/kg)
900
800
700
600
500
400
300
200
100
0
0
20
40
60
80
Silt (%)

e)

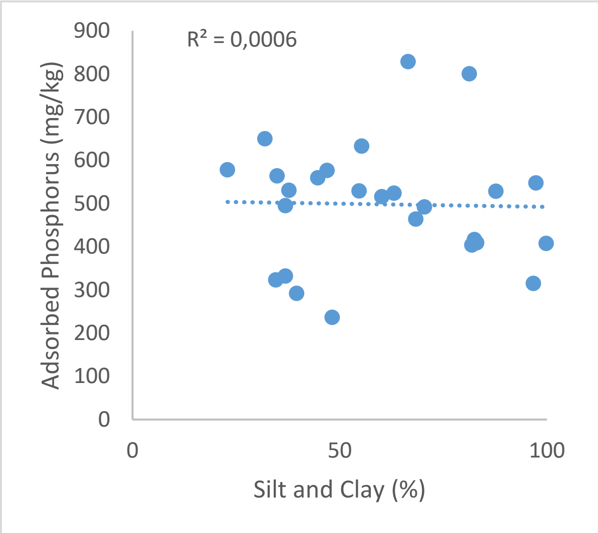


Figure 5.5: Scatter plots displaying the relationship between adsorbed phosphorus and a) orthophosphate concentration in the water, b) percentage of organic matter, c) percentage of clay sediment particles, d) percentage of silt sediment particles and e) the sum of silt and clay.

5.3 Comparison of the concentrations of adsorbed phosphorus and background levels at a natural reference site

The average P concentration measured at the natural 'reference' sites (S2 and S26), i.e., background concentration in the catchment, was found to be 494 mg/kg. Adsorbed phosphorus concentrations in the catchment were compared to background level concentration by calculating the difference between the two. The results are presented in table 5.4 below. The phosphorus levels recorded in the upper reaches of the catchment are all below background levels. The concentrations start exceeding background levels from the middle reaches of the catchment all the way downstream. Sites with the highest differences are S15, S22 and S17.

Table 5.4: Comparison of adsorbed phosphorus concentrations to background levels at a 'reference' site.

Sample ID	P (mg/kg)	Difference	Comparison to background levels
S1	408	86	Lower

S2	409	85	Lower
S3	315	179	Lower
S4	323	171	Lower
S5	332	162	Lower
S6	237	257	Lower
S7	531	-37	Higher
S8	516	-22	Higher
S9	292	202	Lower
S10	404	90	Lower
S11	560	-66	Higher
S12	633	-139	Higher
S13	564	-70	Higher
S14	529	-35	Higher
S15	829	-335	Higher
S16	416	78	Lower
S17	650	-156	Higher
S18	492	2	Lower
S19	529	-35	Higher
S20	576	-82	Higher
S21	495	-1	Higher
S22	801	-307	Higher
S23	548	-54	Higher
S24	578	-84	Higher

The difference of the concentrations of adsorbed phosphorus at the sampling sites and natural reference sites is mapped below (Figure 5.6). Sites that are classified as 'Lower' indicate that the adsorbed phosphorus concentrations at those sites are lower than the average background concentrations while sites that are classified as 'Higher' indicate that adsorbed phosphorus concentrations at those sites are higher than average background levels. Out of the twenty-four sites that were sampled fourteen of them exceeded the background phosphorus levels while ten of them had concentrations that were below background levels. Most of the sites that indicated lower average phosphorus concentrations are distributed in the upper catchment near the source of the river. Higher concentrations were observed in sediment samples that were taken from sites that are in the middle and lower reaches of the catchment.

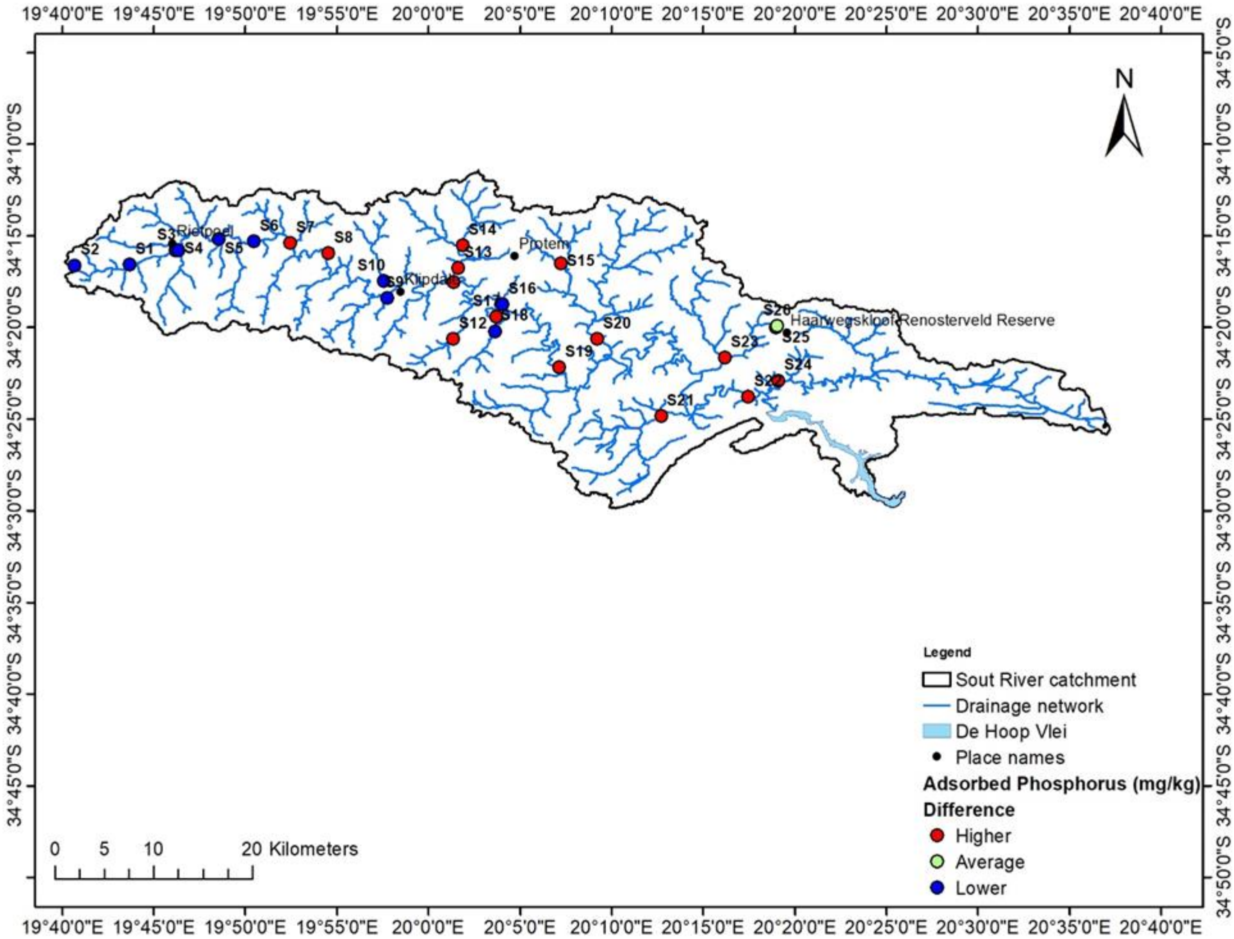


Figure 5.6: The difference of the adsorbed phosphorus concentrations and average background phosphorus levels at the various sampling sites.

5.4 Identification of high-risk areas of adsorbed phosphorus contamination in the Sout River catchment

Figure 5.7 shows the spatial variation of sediment connectivity and adsorbed phosphorus concentrations at the various sampling sites. According to the map, areas that are classified as low risk are associated with lower concentrations of adsorbed phosphorus while areas that are classified as high risk are associated with higher concentrations of adsorbed phosphorus. In general, areas that surround Protem (northern side of the catchment) have the highest risk of contamination. This is because of the high IC values and high phosphorus concentrations. Areas around Klipdale (Southern side of the catchment) have lower contamination risk due to lower IC and phosphorus-bound sediments.

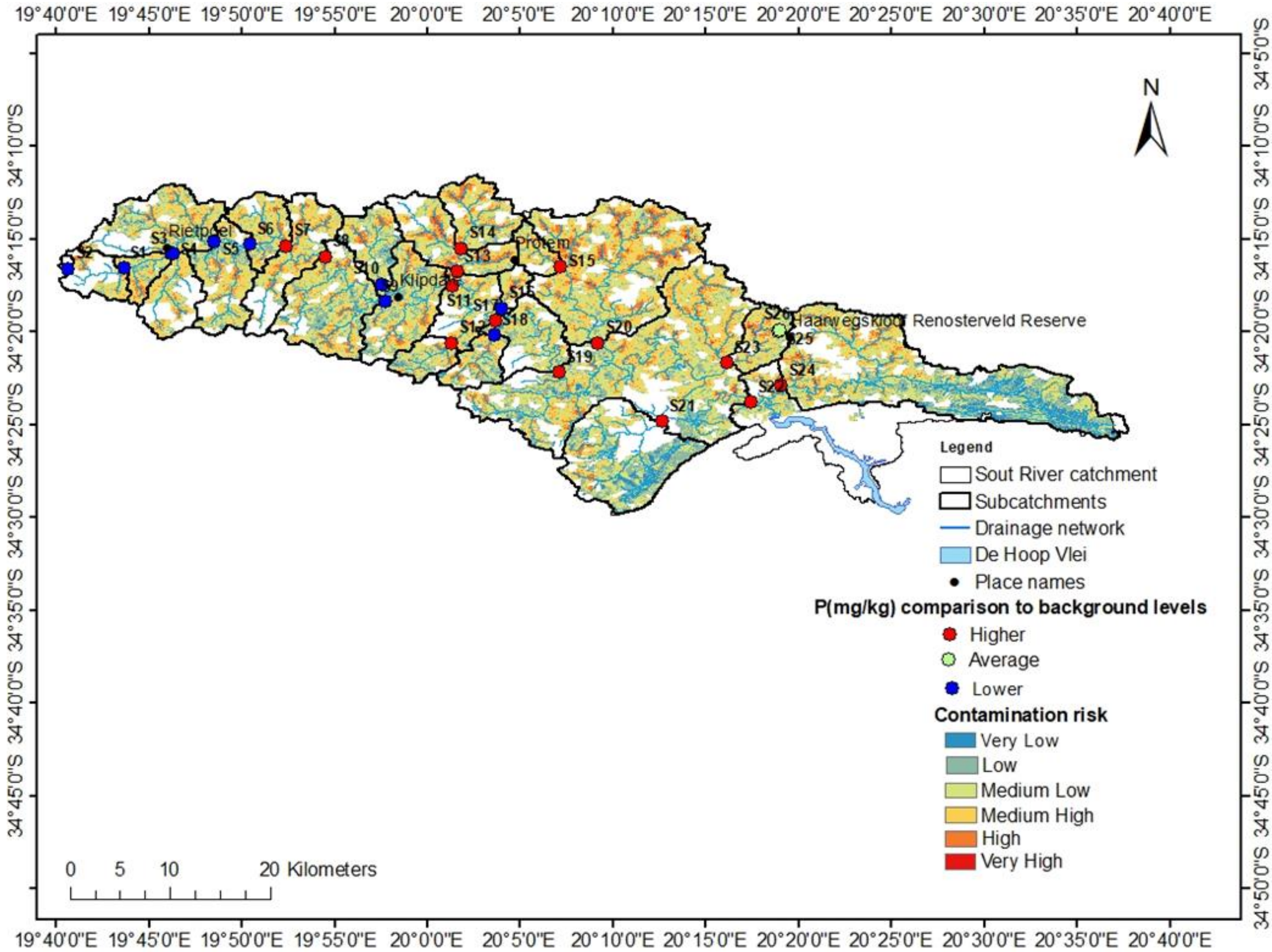


Figure 5.7: Contamination risk associated with adsorbed phosphorus in the Sout River catchment. Higher values indicate high risk while lower values indicate low risk.

Chapter 6: Discussion

6.1 Introduction

The results of this research project have provided an insight into priority areas for riparian buffer restoration in the Sout River catchment. This chapter summarizes the key findings and attempts to explain their implication in terms of the research problem stated earlier in chapter 1 of this study. Study design limitations are also discussed. Lastly, the chapter provides recommendations for future research based on key findings of this research project.

6.2 Determination of sediment connectivity in the Sout River catchment

IC was used to determine sediment connectivity in the catchment to determine potential sediment pathways from hillslopes to the river channel and was found to be high on the northern part of the catchment and low on southern and upper part. The overall IC in the catchment was found to be medium-low due to the lowland nature of the catchment. Vegetation cover, in terms of NDVI, was incorporated into the IC output map to provide more realistic potential sediment pathways in the catchment. Determining sediment pathways is a major step in identifying priority areas for riparian buffer restoration. This is because catchment areas that have high sediment connectivity and low vegetation cover are more susceptible to erosion and can potentially ease the flow of sediments into watercourses. Therefore, it is in such areas that buffer restoration projects should be prioritized.

The integration of IC (Figure 5.1) and NDVI (Figure 5.2) resulted in the risk of sediment contamination map (Figure 5.3) which is used to determine potential sediment pathways into the river channel. The results show high risk of contamination on the northern part of the catchment and most of these areas are along the drainage network. The southern side of the catchment shows low risk of contamination. Overall, there is a medium risk of sediment contamination due to sediment connectivity and vegetation cover in this catchment.

Vegetation cover plays a role in the (dis)connectivity of hillslopes and the main drainage network (Poeppl et al., 2012; Foerster et al., 2014). Since the catchment is dominated by barren land and shrubs and sparse vegetation cover, one would expect this integration to result in an overall increase in the risk of sediment contamination, i.e., increase potential sediment pathways. For example, Grenfell et al. (2022) who modified the weighting factor using the Manning's n to study the effects of a post-fire green flush on target connectivity found that the presence of the green flush decreased the spatial overall connectivity in the Silvermine catchment (Grenfell et al., 2022). However, this was not the case in the present study as the integration of the two maps did not result in any major changes in the potential sediment pathways (Figure 5.3). This may be due to the lowland topography of the catchment and the very low IC values.

Nonetheless, the correspondence of IC and NDVI categories on the maps (low IC corresponds with high NDVI and vice versa) suggests that vegetation is an appropriate measure to be used as a buffer to protect watercourses against sediment contamination. Vegetation acts as a buffer which prevents sediments from entering the drainage network by decoupling hillslopes and the drainage network (Fryirs et al., 2007). Therefore, reducing vegetation cover increases sediment connectivity and may also increase erosion of the topsoil and sediments in the catchment. This is consistent with results obtained by Poeppl et al. (2012) who investigated the influence of riparian cover on diffuse lateral sediment connectivity. The integration of IC and vegetation cover therefore provides an understanding of the (dis)connectivity of the landscape especially in a lowland catchment such as the Sout River catchment where land cover plays a huge role in sediment transport.

6.3 Longitudinal variation in adsorbed P in the Sout River catchment

The natural phosphorus concentration levels in this catchment are relatively high, hence this region is productive for agriculture. However, these concentrations are not equally distributed due to various natural and anthropogenic factors. Therefore, the longitudinal variation in adsorbed phosphorus in the catchment can be used to identify areas in the catchment that are more susceptible to P contamination. These are areas that have high P concentrations compared to background levels and therefore should be prioritized for riparian buffer restoration. The results have shown that concentrations of adsorbed P generally increase longitudinally across the catchment. This is typical of the rivers in the Breede Water Management Area as they have been found to have a generally fair water quality in the headwaters and a declining water quality in the downstream direction (The River Health Programme, 2011).

The upper reaches of the catchment have high elevation (Figure 4.3) which could mean that sediments are easily transported downstream during high flows. In May 2021, the area experienced flooding which may have resulted in the current distribution of phosphorus. Land-use in the Sout River catchment mainly consists of wheat crops and sheep pastures. However, it was observed during sampling that the upper reaches of the catchment were mainly used for crops and no livestock farming was observed at the sampling points (refer to table 2 in the Appendix). The middle and lower reaches of the catchment, on the other hand, consisted of wheat and sheep pastures. This could be another explanation for the low phosphorus concentrations observed in the sediments of these sites.

Since phosphorus adsorbs to sediments, sediment composition influences the concentration of this nutrient which then plays a role in determining its spatial distribution. Previous studies have found that some of the main sediment composition factors that affect phosphorus concentration in sediments are clay and organic matter content. Numerous studies have found that clay particles in the sediments adsorb significant amounts of phosphorus due to their large specific area indicating that sediments with high clay content are likely to have high phosphorus concentrations (Holtan et al., 1998; Reddy et al., 1998; Wang et al., 2009). In this catchment, however, this was not the case as no significant correlations were found between clay particles and phosphorus concentrations, making these finding differ from previous studies.

As explained in Chapter 2, organic matter content can be related to adsorbed phosphorus concentration in sediments. This is because phosphorus tends to be adsorbed into organic matter (Holtan et al., 1988). The present study measured organic matter content of the sediments using the loss on ignition method. The study found no significant correlation between organic matter content and adsorbed phosphorus concentration in this catchment. Similar results were obtained by Yang et al. (2017) where they found that while total phosphorus in some basins was significantly correlated with total organic matter, it was not the case in others. This goes to show the ambiguity of the impact of organic matter on adsorbed phosphorus concentration (Asomaning, 2020). Organic matter inhibits crystallization of iron oxide thereby increasing P sorption; however, it also tends to compete for adsorption sites and therefore decreases P sorption (Asomaning, 2020). This results in contrasting results among studies.

Orthophosphate in the river water was also found to have no relationship with sediment-bound phosphorus in this catchment. This could also be because of the floods; whereby substantial amounts of sediments were deposited in the river channel. This also illustrates that P moves with sediment than with water; they are not in sync. More correlation tests were done to find out what factors determine the spatial distribution of adsorbed phosphorus in the Sout River catchment. These were between adsorbed phosphorus concentration and pH, Al+Fe as well as the electrical conductivity (EC). These results also revealed that there were no significant relationships between these variables and adsorbed phosphorus concentration in this catchment. This also contradicts previous reports that indicated that these variables are associated with the amount of phosphorus that adsorbs to sediments (Holtan et al., 1998; Laakso, 2017; Gupta et al., 2020).

When it comes to nutrient levels and their distribution in a catchment, it is important to not generalize based on findings from other sites because the findings are site specific. The results obtained from this study have demonstrated that phosphorus distribution in a catchment highly depends on site-specific conditions. Therefore, studies must always be done before sediment management decisions can be made.

6.4 Assessment of sediment contamination by comparison to background levels

Adsorbed phosphorus concentrations from various sampling sites in the catchment were compared with background levels from a natural 'reference' site in the same catchment by calculating the difference between the two. This was based on the assumption that sediment concentrations that are not higher than those in the 'reference' site may not be harmful to biota and the environment.

The obtained results show that adsorbed phosphorus levels in the upper part of the catchment are below background levels. This is consistent with what has been previously reported in the Rivers of the Breede Water Management Area whereby the rivers were found to have good quality upstream (The River Health Programme, 2011). The concentrations start exceeding background levels from the middle to the lower reaches of the catchment. This increase may be due to the conversion of the natural vegetation to agriculture. It has been previously reported that the increased availability of soils from agricultural activities has resulted in significant sediment deposition in the Sout River (The River Health Program, 2011). Clearing of the natural vegetation accelerates erosion rates and therefore increases the supply of sediments and associated P to the stream.

6.5 Identification of high priority areas for riparian buffer restoration

The contamination risk assessment map has been developed to identify areas in the catchment that should be prioritized for riparian buffer restoration as the map incorporates both buffer effectiveness (connectivity) and opportunity (phosphorus contamination). The results have shown that the highest connectivity values are on the northern side of the catchment especially in sub-catchments S13, S14, S15 and S20. Interestingly, these sites were also found to have adsorbed phosphorus concentrations that exceed background levels, especially S15 which recorded the highest P concentration. This shows that sediment connectivity in this catchment does influence the distribution of adsorbed phosphorus.

The upper part of the catchment has low to medium connectivity and P concentrations that are below background levels. Areas of high priority for buffer restoration are those that are highly coupled to the drainage network and have high adsorbed P concentrations. In this case, the upper catchment (S1 to S6) is not an area of high priority since it does not meet these requirements. These findings suggest that the combination of buffer effectiveness and connectivity can be used as an effective tool in sediment management.

Chapter 7: Conclusions and recommendations

7.1 Introduction

The research aim of this study is to identify areas in the catchment that should be prioritized for riparian buffer restoration, and this was achieved following these four objectives; to determine sediment connectivity, determine the longitudinal variation in adsorbed phosphorus, compared adsorbed phosphorus concentrations to background levels and finally, identify areas aim the catchment that should be prioritized for riparian buffer restoration. This chapter provides key findings, limitations and recommendations based on the findings of this study.

7.2 Key findings

The IC in the catchment was found to be low in the upper catchment and on the southern side. The highest IC values were mostly on the northern part of the catchment. The NDVI analysis revealed that the catchment is dominated by barren land and sparse vegetation cover, which may be caused by agricultural activities such as livestock and crop farming that dominate the land use in the area. In most cases, agriculture results in the reduction of natural vegetation due to trampling by livestock while grazing and planting of crops. IC and NDVI were incorporated to identify potential sediment pathways, i.e., risk of sediment contamination, due to topography and vegetation cover in the Sout River catchment. Areas with high risk of sediment contamination were found to be on the northern side while some areas in the southern side of the catchment had the lowest risk. The overall risk of sediment contamination in the catchment was found to be medium.

In addition to potential sediment pathways, the study also analyzed the spatial distribution of adsorbed phosphorus to identify areas with high risk of contamination. Sites located on the upper reaches of the catchment recorded the lowest phosphorus concentrations. The concentrations increased downstream and even exceeded background levels. Common sediment composition factors that have been previously reported to be associated with adsorbed phosphorus concentration such as organic matter content and silt and clay content did not have significant correlations with adsorbed phosphorus in this catchment. Additionally, phosphorus concentration in the sediments did not have a significant correlation with orthophosphate in the water. This could be due to the flooding that took place in the area in May 2021.

The present study incorporated buffer effectiveness (sediment connectivity) and opportunity (adsorbed phosphorus concentration) to identify areas that should be prioritized for riparian buffer restoration (Kotze et al., 2009). The study found that areas that are highly connected to the channel are likely to have phosphorus concentrations that exceed background levels. These areas are mostly located on the north-eastern part of the catchment. These are the areas that should be prioritized for riparian buffer restoration since they have a high risk of sediment contamination. In this way, the study was able to establish a link between buffer effectiveness and opportunity. The proposed approach of incorporating buffer effectiveness and opportunity was successfully applied.

7.3 Study limitations

The satellite images and sediment sampling data were acquired far apart due to unavailability of clear high-resolution images that correspond with the sampling dates. Field sampling took place after the area experienced flooding. It is likely that the flood event would have affected sediment distribution within the channel. Due to limited resources and lack of access to private land, sediment sampling was only done once and only a few samples were taken. It is beneficial to sample multiple times to get results that are more accurate.

7.4 Recommendations

The use of a combined approach in identifying riparian restoration priority areas was successfully applied in this catchment. However, the study recommends that future research focuses on studying the factors that control adsorbed phosphorus distribution in this catchment to improve sediment management. Although riparian buffers may work in reducing sediment influx into the river system, they should not be viewed as an end-of-pipe solution as they can get saturated over time. Therefore, it is crucial to control the nutrient at its source.

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APPENDIX

Table 1: Description of the sampling sites

Sample ID	Latitude	Longitude	Site description	Land use
S1	-34,27585	19,72772	Sout, large vlei near source	Valley-bottom wetland, hillslope wheat
S2	-34,27673	19,67767	Source of Sout upstream of S1	Wheat
S3	-34,26325	19,76973	Sout near R326 crossing	Wheat
S4	-34,26308	19,77210	Groenbergkloof tributary near R326 crossing	Wheat
S5	-34,25248	19,80899	Sout at Langkuil	Wheat
S6	-34,25494	19,84088	Sout at Langkuil 2	Wheat and flax
S7	-34,25611	19,87369	Sout at Soutkuil	Wheat and pasture
S8	-34,26583	19,90831	Sout at Graauwheuwel	Wheat
S9	-34,30624	19,96201	Sout at Klipdale	Wheat, flax, light industry
S10	-34,29073	19,95884	Blaauwheuwel tributary	Wheat
S11	-34,29173	20,02277	Sout at Kykoedie (old flow gauge)	Wheat
S12	-34,34340	20,02217	Hermanusheuwel tributary	Wheat
S13	-34,27909	20,02686	lower Hotnotskraal tributary	Wheat, pasture
S14	-34,25822	20,03072	upper Hotnotskraal tributary	Wheat, pasture
S15	-34,27485	20,12001	Witkilp se loop tributary near Langhoogte Dairy	Wheat, pasture, hay

S16	-34,31198	20,06718	Klipbankskloof tributary (turbid, trampling by cattle)	Wheat, pasture
S17	-34,32363	20,06146	Sout at Klipbankskloof	Wheat, renosterveld on steep coupling zones
S18	-34,33664	20,06004	Koeranna se loop tributary (impounded by road)	Wheat, pasture, flax
S19	-34,36945	20,11873	Sout at Welgegund	Confined, renosterveld coupling zones
S20	-34,34341	20,15329	Soerivier (Soes) tributary	Renosterveld coupling zones, wheat/pasture on hills
S21	-34,41397	20,21131	Waterskilpadsrivier tributary	Valley-bottom wetland and panveld, hillslope wheat
S22	-34,39643	20,29062	Sout upstream of De Hoopvlei	Large renosterveld coupling zones, mixed wheat, pasture, renosterveld hills
S23	-34,36084	20,26934	Brakkuil se loop	Semi-natural setting downstream of Haarwegskloof but still some wheat on slopes
S24	-34,38154	20,31802	Potbergsrivier tributary	Large areas of intact renosterveld, some wheat
S25	-34,33329	20,31583	Haarwegskloof valley head ‘reference’ site, left valley fork	Renosterveld, no surface water

S26	-34,33169	20,31682	Haarwegskloof valley head 'reference' site, right valley fork	Renosterveld, no surface water

Table 2: Sediment composition data

Sample ID	Clay%	Silt%	Clay and silt%	Loss on ignition%	P (mg/kg)	Al and Fe (mg/kg)
S1	40,46	59,36	99,81	6,87	408	95842
S2	28,74	54,29	83,03	6,75	409	57285
S3	48,04	48,74	96,78	7,14	315	51014
S4	11,62	22,97	34,59	4,04	323	61768
S5	8,41	28,53	36,94	2,86	332	61259
S6	10,96	37,22	48,18	2,32	237	39744
S7	3,29	34,49	37,78	3,46	531	45750
S8	19,41	40,80	60,21	4,99	516	62980
S9	9,44	30,18	39,63	2,76	292	67088
S10	30,10	51,79	81,89	8,80	404	72462
S11	15,31	29,43	44,74	4,60	560	62148
S12	22,26	33,03	55,29	9,96	633	76793
S13	10,81	24,14	34,94	5,84	564	64661
S14	39,55	48,18	87,73	19,36	529	64168
S15	22,47	44,04	66,51	6,59	829	62398
S16	29,11	53,38	82,49	4,88	416	68131

S17	10,92	21,04	31,96	5,43	650	49356
S18	19,01	51,43	70,45	8,77	492	78536
S19	21,18	33,49	54,67	3,44	529	52262
S20	13,84	33,15	47,00	4,62	576	63208
S21	10,86	26,03	36,88	7,55	495	43873
S22	33,56	47,70	81,26	11,02	801	71290
S23	45,42	51,97	97,39	9,22	548	110915
S24	4,47	18,48	22,95	2,00	578	45103
S25	14,69	48,46	63,14	37,74	524	39068
S26	18,09	50,31	68,40	8,75	464	66950

Table 3: Water samples composition data

Sample ID	PO_4 (mg/L)	Lab Temperature (°C)	EC	pH
S1	0,8	17	3,19	8,44
S2	0,9	17	7,25	8,31
S3	0	17	30,44	8,73
S4	1,9	17	9,81	9,40
S5	1,6	17	10,40	9,15
S6	1,1	17	9,95	8,95
S7	0,7	17	8,84	8,85
S8	0,5	17	8,19	8,91
S9	0,2	17	7,48	8,95
S10	0	17	9,17	9,00

S11	0,2	17	8,69	8,86
S12	1,6	17	34,91	8,81
S13	0	17	8,86	9,02
S14	1,6	17	10,08	8,76
S15	0,6	17	6,03	9,06
S16	1,5	17	8,43	9,15
S17	0	17	9,18	9,02
S18	0,1	17	29,60	9,17
S19	0,1	17	8,70	9,28
S20	0	17	13,42	8,93
S21	1	17	34,43	8,68
S22	0	17	12,59	8,90
S23	0	17	15,64	8,88
S24	0	17	13,51	8,62
S25	no water	no water	no water	no water
S26	no water	no water	no water	no water