THE USE OF RIPARIAN BUFFER ZONES FOR THE ATTENUATION OF NITRATE IN AGRICULTURAL LANDSCAPES

by

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ABSTRACT

The focus of this mini-dissertation is the use of riparian buffer zones to manage nitrate pollution of water resources. Riparian buffer zones are vegetated areas adjacent to streams, lakes and rivers, that are managed to enhance and protect aquatic resources from the adverse impacts of agricultural practices. These zones are recognised globally for their function in water quality amelioration. Despite the growing literature, there is little consensus on how to design, assess and manage these riparian buffer zones specifically for nitrate attenuation.

For the purpose of this mini-dissertation, a literature review of world-wide research into the nitrate attenuation efficiencies of riparian buffer zones was undertaken. A database was created using the key information from this literature. Two key processes responsible for immobilising and/or removing nitrate from surface and subsurface flows are generally recognised in the available literature, namely: vegetative uptake and the process of denitrification. A comparison of the available riparian studies indicated that there are similar characteristics in riparian buffer zones that may be responsible for enhancing these key mechanisms. Studies where there was shallow lateral subsurface or uniform surface water delivery pathways, vegetation of close structure and composition, high organic matter in the soils and fluctuating soil surface saturation rates showed the most significant nitrate attenuation efficiencies.

The mini-dissertation proposes that these similarities can be used to both assess a riparian landscape for its potential to attenuate nitrate, and to size a riparian buffer zone specifically to meet this function. A set of proposed guidelines based on the findings of the dissertation attempt to illustrate how riparian pollution control recommendations can be achieved. These guidelines are an example of how to assist a farmer or similar landowner in achieving good nitrate removal efficiencies from a riparian buffer zone. The guidelines work through three steps, which help to establish and prioritise management zones, assess each zone 's potential for nitrate attenuation, and determine adequate riparian buffer widths for each management zone. A case study was used to illustrate the practical application of the guidelines. Full testing of these guidelines was not within the scope of this mini-dissertation, however the guidelines are an indication of how information regarding riparian function can be applied to a system to determine effective management of water resources.

PREFACE

The research described in this mini-dissertation was carried out at the Centre for Environment and Development, University of Natal, Pietermaritzburg, under the supervision of Dr Nevil Quinn (Centre for Environment and Development), and was funded by the Water Research Commission.

The mini-dissertation represents the original work of the author and has not otherwise been submitted in any form for any degree or diploma at any university. Where use has been made of the work of others it is duly acknowledged in the text.

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1. INTRODUCTION

1.1 SOUTH AFRICAN WATER SYSTEMS AND MANAGEMENT

Water quality and quantity are of fundamental importance to the continued sustainability of freshwater resources, and the existence of ecosystems and human populations reliant on these resources. As South Africa is a country dedicated to improving the health and living standards of its people, the importance of water quality and quantity in this country is enhanced. This country recognises the interrelated nature of environment and development, and aspires to achieve these goals in an environmentally sustainable and equitable manner. Water of great quantity is of little use to many water users and natural riverine ecosystems if it is of poor quality. Poor water quality negatively affects all water users, such as those relying on water for consumption, agriculture, industry, conservation or recreation. Improved water quality will not only impact on these water users, but also produce non-user benefits associated with securing the sustainability of the aquatic environments and their related ecosystems. Improving the water quality in South Africa is necessary if better health and living standards are to be achieved.

Readily available freshwater is a limited resource in South Africa. The growing demand for freshwater resulting from rapid urbanization and increasing water pollution is of concern, and has attracted the attention of water users, water managers and providers, and environmentalists. Natural permanent bodies of standing freshwater are almost non-existent in South Africa (Dallas & Day 1993). South African dams have large surface areas to compensate for their limited depth due to the nature of the country's topography. The problems and implications of damming, coupled with high evaporation rates, make this form of water storage difficult. Rivers therefore remain one of the few exploitable sources of freshwater in this country (O'Keeffe *et al.* 1998).

Although there are numerous South African rivers, many of these are non-perennial and short in length. Rivers are particularly susceptible to pollution as they are uni-directional drainage systems (Van der Merwe & Grobler 1990). South African rivers are commonly associated with poor water quality, excessive eutrophication (aquatic plant growth) and unhealthy ecosystems often resulting from agriculturally derived sedimentation and pollution (Giliomee 1998). Nitrate is one of the most commonly recognised agricultural related pollutants of water resources, which is usually applied to croplands as fertilizer. It is implicated as one of the main compounds contributing to excessive eutrophication of many South African rivers and dams (Dallas & Day 1993). Both marine and freshwater eutrophication can have negative effects on aquatic and terrestrial ecosystems, and may be detrimental to the health of humans. Excessive aquatic weeds, algae and zooplankton may interfere with water utilisation such as consumption, recreation, subsistence and commercial farming, and industry (Carpenter 1998) (Section 2.5.1).

The problems of nitrate pollution, coupled with the limited water resources in South Africa, require careful water planning, management and conservation guidelines based on scientific principles. The need for sustainable and equitable water utilisation has also been recognised at a global scale, where it has become a major policy priority for most countries to incorporate characteristics of sustainabilty and equity into water planning and policy goals (Gleick 1998; Dowdeswell 1998).

The most recent response to the degraded state of most of South Africa's water systems was the creation of a new water policy and law, the National Water Act 36 of 1998 (NWA). The NWA has been considered globally as one of the most advanced and comprehensive pieces of legislation regarding the control and management of water. The Act redefines appropriate water rights and uses, with implications for people throughout the country. The NWA sets out specific guidelines regarding the control, management, utilisation and conservation of South Africa's water resources. Sustainability and equity are identified as central guiding principles.

The NWA attempts to delegate water resource management to the regional or catchment level through the establishment of Catchment Management Agencies (CMA's). These agencies are to represent individual water management areas. One of the initial functions of the CMA's will be to develop Catchment Management Strategies (CMS's) for their water management areas. A CMS is a proposed strategy for the protection, use, development, conservation and management of the water management area. A purpose of the NWA is to reduce and prevent pollution and degradation of water resources. CMA's are empowered by the NWA to conserve and protect the water resources and resource quality within the management area. Agencies such as these will be able to implement programmes targeting problems such as agriculturally derived nitrate pollution of water resources.

Although this legislation exists, specific water management programme guidelines to address problems such as agriculturally based pollutants are still lacking. Examples of such programme guidelines can be found in America, Australia, Europe and New Zealand. Australian land and water-use industry consultants and agencies are supported by institutes such as the Cooperative Research Centre for Catchment Hydrology (CRC). Initiatives of the CRC promote the transfer of technology and expertise, and the advancement of water catchment research.

One such initiative is the CRC's Waterway Management Programme, aimed to control the delivery of sediment and nutrients in water supply catchments, as well as river channel and bank stability through specific guidelines. The riparian ecosystem management model (Altier *et al.* 1999) has been widely adopted in the United States, and is another example of management guidelines for agriculturally based pollutants.

Riparian buffer zones are vegetated areas adjacent to streams, lakes and rivers, that are managed to enhance and protect aquatic resources from the adverse impacts of agricultural practices (Dosskey *et al.* 1999). International experience supports the use of riparian buffer zones for the specific function of water quality amelioration. For example, in 1996, members of the Chesapeake Executive Council, Virginia, USA, adopted the Riparian Forest Buffer Initiative. This initiative aims to preserve, protect and enhance existing forested buffers, and aspires to plant an additional 2010 miles of stream side buffers in Chesapeake Bay.

1.2 PROBLEM STATEMENT

The focus of this mini-dissertation is the use of riparian buffer zones to manage nitrate pollution of water resources. Riparian buffer zones have been recognised as the most successful attenuation mechanism for agriculturally derived non-point source nitrate (Hill 1996). Although much research has focussed on the mechanisms of nitrate attenuation in riparian buffer zones, there still lacks sufficient guidelines towards the sizing and management of the zones specifically for the function of nitrate attenuation.

Riparian buffer zones have proved both very efficient (greater than 94% of the nitrate removed) (Peterjohn & Correll 1984), and very inefficient (less than 15% of the nitrate removed) (Cooper 1990) at nitrate attenuation. The variation in these findings may be attributed to the design of the riparian buffer zones, or other on-site differences. Recommendations regarding the minimum riparian buffer width for achieving good nitrate attenuation range from five metres to ninety metres (Section 2.6.5). This suggests that if nitrate attenuation is the management objective, then standardised guidelines for the design and establishment of these riparian zones specifically for this function are necessary.

This mini-dissertation attempts to establish the existing knowledge about agricultural non-point source nitrate pollution, and its attenuation by riparian buffer zones. An understanding of the riparian buffer zone water quality functions and mechanisms responsible for nitrate removal can assist in the design, establishment and management of riparian buffer zones to enhance this function. The mini-dissertation looks towards management recommendations for riparian buffer zones by suggesting possible guidelines for assessing these zones and sizing them for improved nitrate removal. It is anticipated that a smaller variation in width recommendations may be achieved through the understanding of commonalities in riparian buffer zones responsible for improving nitrate attenuation.

1.3 AIM AND OBJECTIVES

The research is aimed at promoting the use of riparian vegetation as part of farming practice, and at developing guidelines that will facilitate recommendations regarding the establishment and management of riparian buffer zones specifically for the removal and immobilisation of nitrate, particularly within an agricultural context. The aim of this mini-dissertation requires that the following objectives are met:

i) the identification of the mechanisms responsible within riparian buffer systems for the retention and removal of nitrate

ii) a review of the relevant studies, and isolation of the characteristics of riparian buffers responsible for enhancing the mechanisms of nitrate attenuation

iii) developing a means of assessing riparian areas and sizing riparian buffer zones specifically for the function of nitrate attenuation

iv) the use of a case study to indicate how the buffer assessments are to be used.

1.4 METHODS

The methods of this mini-dissertation are inductive, and are based primarily on a literature review, with limited testing of the proposed approach. The literature review was undertaken in order to establish what is known about the water quality amelioration functions of riparian land environments. The broad theoretical framework of riparian buffer function was documented, and uncertainties and discrepancies in riparian buffer zone knowledge noted (Chapter 2). The broader topic of sediment and nutrient retention functions of riparian land environments indicated that nitrate pollution in South African water systems required closer attention. This resulted in the focus on nitrate removal or immobilisation within riparian buffer zones.

The available studies and experiments relating to the topic were collated, and entered into a data base system. The purpose of this system was to compare available data, and isolate characteristics promoting nitrate removal and retention within riparian buffer systems. Manipulation of the data in the form of graphs was required to better analyse these comparisons (Chapter 3). The buffer zone characteristics and commonalities found to be most responsible for promoting nitrate removal or immobilisation were then used to develop an example of a means of assessing nitrate attenuation in any given landscape. Such an assessment will allow the user to identify which of these characteristics are present or absent within their existing riparian system. Once the site potential for nitrate attenuation is established it is possible to design a site-specific riparian buffer system of suitable width and composition for good attenuation of nitrate. A set of guidelines were developed as based on riparian commonalities promoting nitrate attenuation (Chapter 4). These guidelines are proposed as a means of assisting riparian buffer zone assessment and design for nitrate removal or retention. The guidelines for the assessment and design of riparian buffer systems for nitrate attenuation were further developed through practical application in a case study. The study site chosen is a potato, maize and dairy farm (Sourveld Farm) situated in the Kamberg-Kangatong area, west of Nottingham Road in the KwaZulu-Natal Midlands.

1.5 DOCUMENT STRUCTURE

The first chapter aims to provide a rationale and purpose of the research, and outlines the general objectives of the dissertation. Chapter 1 also discusses the methodology, with a description of the way in which the topic was approached. Chapter 2 explores the general value of riparian environments as sediment and pollutant filters and traps. The general agricultural pollutants of particular concern to water management include those of nitrogen. This chapter describes the nitrogen cycle within an agricultural context, and discusses various forms of nitrogen important to the focus of this dissertation. The sources and consequences of nitrogen pollution are also identified, with a focus on nitrate in the agricultural landscape. Chapter 3 explores the key mechanisms of nitrate attenuation in riparian buffer zones, and provides an understanding of how these key mechanisms function. This chapter makes use of world-wide literature and studies of nitrate in riparian buffer zones, and identifies commonalities and conditions important to the key mechanisms of nitrate retention and removal. These commonalities are used in Chapter 4 to develop an example of riparian buffer zone assessment and width determination for the improved function of nitrate retention and removal. A case study was used as a practical approach to the development of the assessments, and these findings are documented in the fourth chapter. This chapter looks towards riparian buffer zone management recommendations. Chapter 5 reflects on the general findings of the dissertation in the form of a discussion. The dissertation is brought to closure in the conclusion in Chapter 6, and recommendations are made regarding the potential future research opportunities that have arisen through the exploration of this mini-dissertation topic.

2. NITROGEN IN AGRICULTURAL LANDSCAPES

2.1 INTRODUCTION

Management of excessive nitrogen in agricultural landscapes requires a greater understanding of its origin, travel and consequences to aquatic systems. This chapter aims to explore the nitrogen cycle within an agricultural context, with particular focus on solute-nitrogen in the form of nitrate. The chapter investigates the use of riparian buffer zones as a solution to the consequences of excessive nitrogen pollution to aquatic resources.

2.2 AGRICULTURAL LANDUSES AND RIPARIAN ENVIRONMENTS

A riparian area can be described as the part of a landscape adjoining streams and rivers, which directly influences, or is influenced by the aquatic margins and the diversity of ecosystems and biota contained within these margins. The National Water Act (36 of 1998) defines a riparian habitat as "the physical structure and associated vegetation of the areas associated with a watercourse which are commonly characterised by alluvial soils, and which are inundated or flooded to an extent and with a frequency sufficient to support vegetation of species with a composition and physical structure distinct from those of adjacent areas". The word "riparian" refers to land adjacent to a body of water, as it is derived from the Latin word "*ripa*" meaning bank or shoreline (Gold & Kellog 1997; Palone & Todd 1997). The environment may include streambanks, flood plains, and wetlands, and the transitional zone of sub-irrigated sites between the aquatic and terrestrial environments. Trees, shrubs and grasses may vegetate the riparian environment, and are able to intercept contaminants from both surface and subsurface water before stream entry (Dallas & Day 1993; Castelle *et al.* 1994; Hill 1996; Correll 1997; Karssies & Prosser 1999).

A diversity of riparian environments have been defined in legal, biological, botanical, geographical and functional contexts, with the choice of definition dependent on the management aim or situation. Riparian areas have been defined by their uses in legal contexts, by their soil and hydrological or plant community characteristics in biological contexts, and by their functions in conservation from an ecosystem perspective (Table 2.1).

Legislative definitions of set riparian buffer width may not consider individual stream factors (such as channel size and shape, flow characteristics and contributing catchment) which may influence the extent of the riparian area. Those definitions which may consider the immediate characteristics of the river channel in question (such as landform definitions), may neglect to consider the adjacent features such as wetlands and estuaries, which will also affect the extent of the riparian area. Definitions based on distinct vegetation type and immediate site characteristics (such as vegetation definitions) are often impractical as riparian areas are so varied both between sites and within sites. For example, a single riparian area may have a range of vegetation types, from well established mature trees to emergent macrophytes (LWRRDC 2000). Functional definitions attempt to consider all these factors by defining a riparian area in terms of it's functions and benefits, and as such are usually the most widely adopted choice of definition.

Table 2.1: Definitions as based on context in which riparian areas are used/managed (Karssies & Prosser 1999; LWRRDC 2000).

CONTEXT	DEFINITION
LEGAL	Riparian land constitutes a set width (usually 20m - 40m, according to the Act or country in which the area is defined) along the banks of designated rivers and streams.
BIOLOGICAL	**Vegetation**: riparian lands can be distinguished by the vegetation which is obviously (often visually) different to the surrounding terrestrial land. **Landform**: that area between the low-flow level of the watercourse and the highest point of the transition between the channel and its floodplain.
FUNCTIONAL	Riparian lands are part of the landscape adjacent to streams which exert a direct influence on streams or lake margins and on the water and aquatic systems contained within them. The definition may be accompanied by an indication of: - the type of features directly affected by the riparian area including the channel morphology and bank stability, - the physical and chemical properties of the water, - the aquatic ecosystem and water quality, - the conservation; recreation; aesthetic; or commercial values of the given riparian area in question.

As riparian environments are so varied, not only between and within regions, but between and within catchments too, it is difficult to quantify the boundary specifications within any one definition. It is generally accepted however, that riparian areas include the streambanks and a variable size belt of land alongside these banks (Karssies & Prosser 1999).

Riparian areas are responsible for providing important aquatic and terrestrial habitats, controlling stream temperature and bank stability; and maintaining stream flow (Figure 2.1). In a review of riparian forest buffers by Klapproth (1999) it was found that one of the most important functions of riparian systems is the protection of the stream ecosystem through the improvement of water quality entering the environment. Vegetation within a riparian environment is considered extremely efficient in reducing the velocity of water flow entering a stream system. Sediment, nutrients and attached pollutants contained in both surface runoff and subsurface flow, are deposited, trapped or utilised within the riparian area (Osborne & Kovacic 1993; Barling & Moore 1994; Karssies & Prosser 1999). Thus the vegetation acts as a natural filter system, influencing quantity of sediments, nutrients, salts and other contaminants entering the stream.

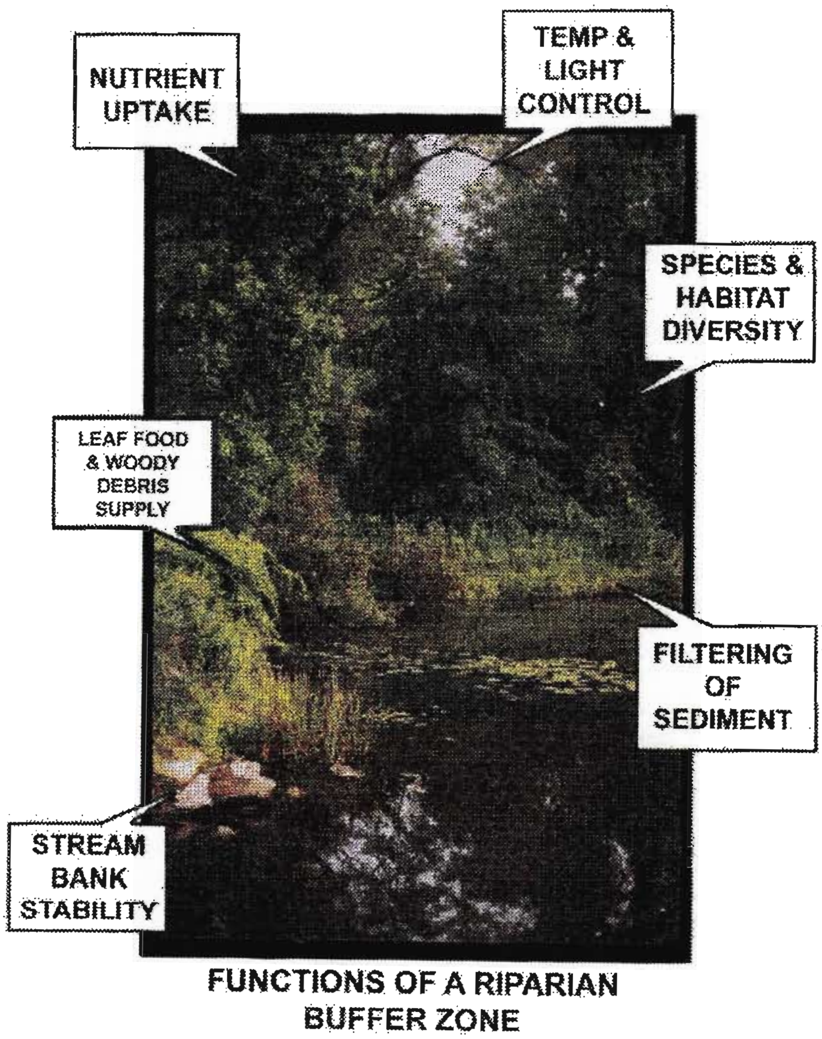


Figure 2.1: The variety of functions that a riparian environment can fulfill

Riparian shorelines are usually significantly valued properties in terms of development. Common land uses of riparian areas include cultivated agriculture, pasture, recreational facilities, forestry, and residential, commercial or industrial development. Riparian corridors become fragmented by the landuse practices, resulting in losses in connectivity and complexity

(Wissmar & Beschta 1998).

The fertile floodplain soils of riparian environments are highly sought after for agricultural landuses. Agricultural land use is a human activity which alters the natural ecosystems such that yield of plant and animal products desired is increased. The diverse landuse patterns in South Africa reflect the diversity in the country's climate, soils and topography, and result in the production of a wide range of agricultural commodities for domestic and foreign markets. About 83% of the land surface in South Africa is utilised by the agricultural sector (CSIR 1992), where at least 70% of the country is designated natural grazing land; and 13% of the country is under cultivation (Schoeman & Scotney 1987).

The agricultural sector in South Africa has monopolised the water resources through irrigation and stock watering, where in 1965 over 83% of the countries total water consumption was as a result of agricultural activities. This figure has more recently been reduced where the agricultural sector now only utilises 67% of the country's direct water use (Rabie & Day 1998). Both the quality and quantity of water in South African rivers have been affected by impoundments; excessive abstraction; catchment mismanagement; alien vegetation infestation; salinisation; eutrophication; and sedimentation from agricultural production (DWA 1986).

The productivity potential of riparian environments often results in the overutilisation of these areas for agricultural purposes. Uncontrolled clearing, intensive cropping, overgrazing and intensive irrigation practices are commonly associated with these riparian environments, and may cause degradation or deterioration of riparian systems and functions (NRCS 1997b). The increasing nutrient and chemical inputs to South African river systems and water resources (Giliomee 1998) may be partially as a result of the degraded state of riparian areas.

2.3 AGRICULTURAL PRACTICES AS A NUTRIENT SOURCE

Nutrients are essential elements for aquatic ecosystems. Normal plant growth and reproduction are dependent on many plant nutrients such as nitrogen, phosphorous, potassium, carbon, calcium, sulphate, silica, magnesium and micro-nutrients. Generally, these nutrients, even in high concentrations are not toxic, but in excessive amounts may alter aquatic environments and reduce water quality (Dallas & Day 1993; Klapproth 1999). High concentrations of nutrients may have a significant impact on the structure and functioning of biotic communities within the aquatic ecosystems.

The natural provision of nutrients to a particular aquatic system are relatively constant (Dallas

& Day 1993). Factors contributing to this natural nutrient provision include climatic factors and catchment characteristics. Natural nutrient inputs into surface waters, such as those derived from plant materials and eroding soils, are often increased by nutrients derived from human activities (Figure 2.2). Agricultural practices, industrial sources and waste management are some of the activities contributing to increased inputs of nutrients.

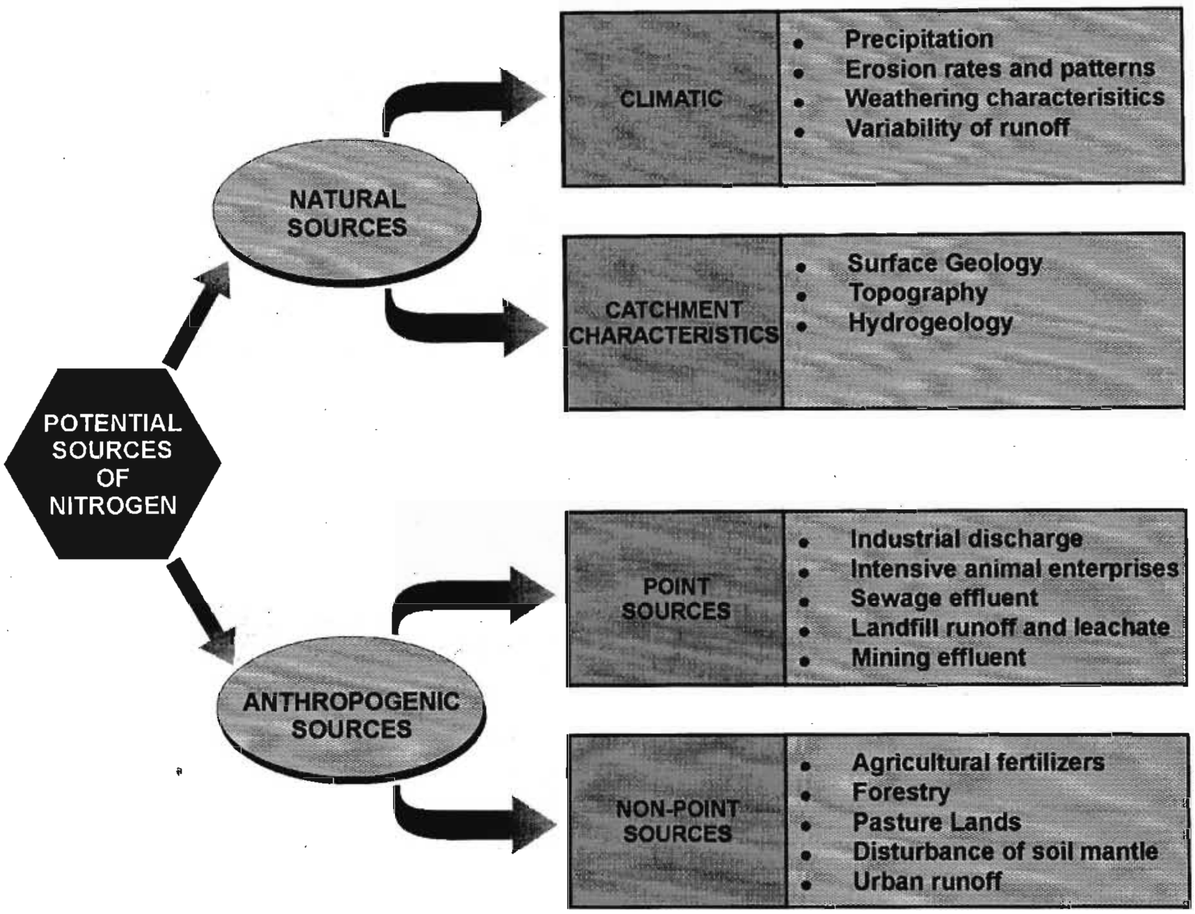


Figure 2.2: Potential sources of nutrients to agricultural landscapes (after Dallas & Day 1993)

One of the most significant water quality problems facing much of the world is non-point source pollution from agricultural areas (Dillaha & Inamdar 1997). In the United States, non-point source pollution prevention is a national priority. Over 30% of the US waters do not fulfill their designated purposes, and two thirds of the problems associated with these impaired waters can be related to non-point source pollution (Altier *et al.* 1999). The agricultural sector is named as one of the major contributors to non-point source pollution in the USA. Dillaha and Inamdar (1997) report that in 1995 the United States Environmental Protection Agency identified agriculture as the primary source of pollution to rivers and lakes, and the main source of sediment in all water bodies.

2.4 A FOCUS ON NITROGEN

Nutrient transport by farming systems of the world's agricultural lands has overwhelmed natural nutrient cycles, and it is thought that more nutrients from fertilisers enter the cycle than are removed as produce (Carpenter *et al.* 1998). Unnaturally high concentrations of nitrogen and phosphorous from fertilisers are most commonly responsible for eutrophication (nutrient enrichment) of aquatic systems, and the resultant excessive plant growth (Dallas & Day 1993). Phosphorous is more widely implicated as the primary cause of eutrophication in systems. This element may be part of bacteria, fungi, algae and other organisms, but may also enter aquatic environments attached to soil particles, or in organic materials in surface runoff after rainfall events. Nitrogen, however, is increasingly being recognised in other parts of the world for its primary responsibility in eutrophication of temperate estuaries and coastal ecosystems, as it is the limiting factor in the primary production of these plant habitats (Carpenter *et al.* 1998). Organic nitrogen, like phosphorous, enters aquatic systems attached to sediment, but inorganic nitrogen and ammonium ions enter as solutes.

Nitrogenous fertilisers are usually the most important fertilisers applied to plants, as nitrogen is often the primary limiting nutrient in many plant communities (Barbour *et al.* 1987). Weier *et al.* (1982) list the chief nitrogenous fertilisers as those with nitrogen in the form of: a) nitrate; b) ammonia or its compounds; c) organic compounds, such as cottonseed meal; and d) amide, such as urea and calcium cyanamide.

Nitrogen containing fertilisers and fossil fuels are often considered pollutants as they release excess nitrogen into the system. Isermann (1991 cited in Carpenter *et al.* 1998) states that only 18% of the nitrogen input in fertiliser is removed from United States farms as produce, leaving an average of 174 kg/ha/yr of surplus nitrogen. Intensive animal productions often recycle nutrients through application of manure to croplands, however manure yields from concentrated livestock productions often exceed the capacity of croplands to sequester nutrients (NRC 1993b cited in Carpenter *et al.* 1998). The surplus nitrogen available from fertiliser and manure accumulates in soils, erodes or leaches into surface and subsurface waters, is transported to aquatic ecosystems, or enters the atmosphere.

2.4.1 The nitrogen cycle

Nitrogen, together with oxygen, carbon and hydrogen, is one of the four most common elements in living cells, and an essential constituent of proteins and nucleic acids, which are two groups of substances largely responsible for supporting life (Tamm 1991). Proteins include enzymes which catalyse all biochemical processes, and are therefore components of all living organisms, whilst nucleic acids (DNA and RNA) determine the pattern for an organisms growth and development (Dallas & Day 1993). Practically all biochemical processes and biological structures involve nitrogen-containing compounds.

In order to better identify a means of controlling or preventing increased nitrogen content within aquatic systems, it is first necessary to understand the nitrogen cycle (Figure 2.3). Nitrogen makes up 78% of the troposphere, but as nitrogen gas (N_2) occurs combined in stable diatomic molecules and resists uniting with any other common substances, it remains inert. Nitrogen therefore cannot be used directly as a nutrient by multicellular plants or animals in its gaseous form. The strong covalent bond associated with nitrogen gas can only be broken by lightning, volcanic action and nitrogen fixation, allowing nitrogen to be removed from the atmosphere and converted into compounds that are able to enter food webs (Barbour *et al.* 1987).

Table 2.2: Description of five processes of the nitrogen cycle as seen in agricultural landscapes

	NITROGEN CYCLE: DEFINITIONS
1. NITROGEN FIXATION	Conversion of gaseous nitrogen to ammonia by specialized bacteria found in plants and the soil.
2. NITRIFICATION	Two-step process where ammonia is converted to nitrite ions, and then to nitrate ions.
3. ASSIMILATION	Absorption of nitrate ions, inorganic ammonia, and ammonium ions by the roots of plants.
4. AMMONIFICATION	Conversion of nitrate back to nitrite, and then to ammonia and ammonium ions by fungi and bacteria.
5. DENITRIFICATION	Conversion nitrate to gaseous nitrogen to be released into the atmosphere.

Nitrogen fixation may be both natural (by nitrogen-fixing bacteria), or a result of anthropogenic processes such as the Haber Process in industry (Postgate 1978). Nitrogen fixation refers to the process whereby specialized bacteria (eg: cyanobacteria in the soil and water, and Rhizobium bacteria in the nodules of legumas plants) convert gaseous nitrogen (N_2) to ammonia (NH_3) by the reaction:

$$N_2 + 3H_2 \rightarrow 2NH_3$$

Ammonia and ammonium ions (NH_4^+) (formed when ammonia reacts with water) can be used by plants as a source of nitrogen. Specialized aerobic bacteria in the soil convert ammonia to nitrite ions (NO_2^-) which are toxic to plants, and then to nitrate ions (NO_3^-) which are easily available to plants as a nutrient. This two-step process is referred to as nitrification. Plant roots absorb nitrate ions, inorganic ammonia, and ammonium ions by the process of assimilation. Nitrate ions are used by the plants in the biosynthesis of amino acids, proteins and nucleic acids. Animals in turn obtain their nitrogen through the consumption of plants.

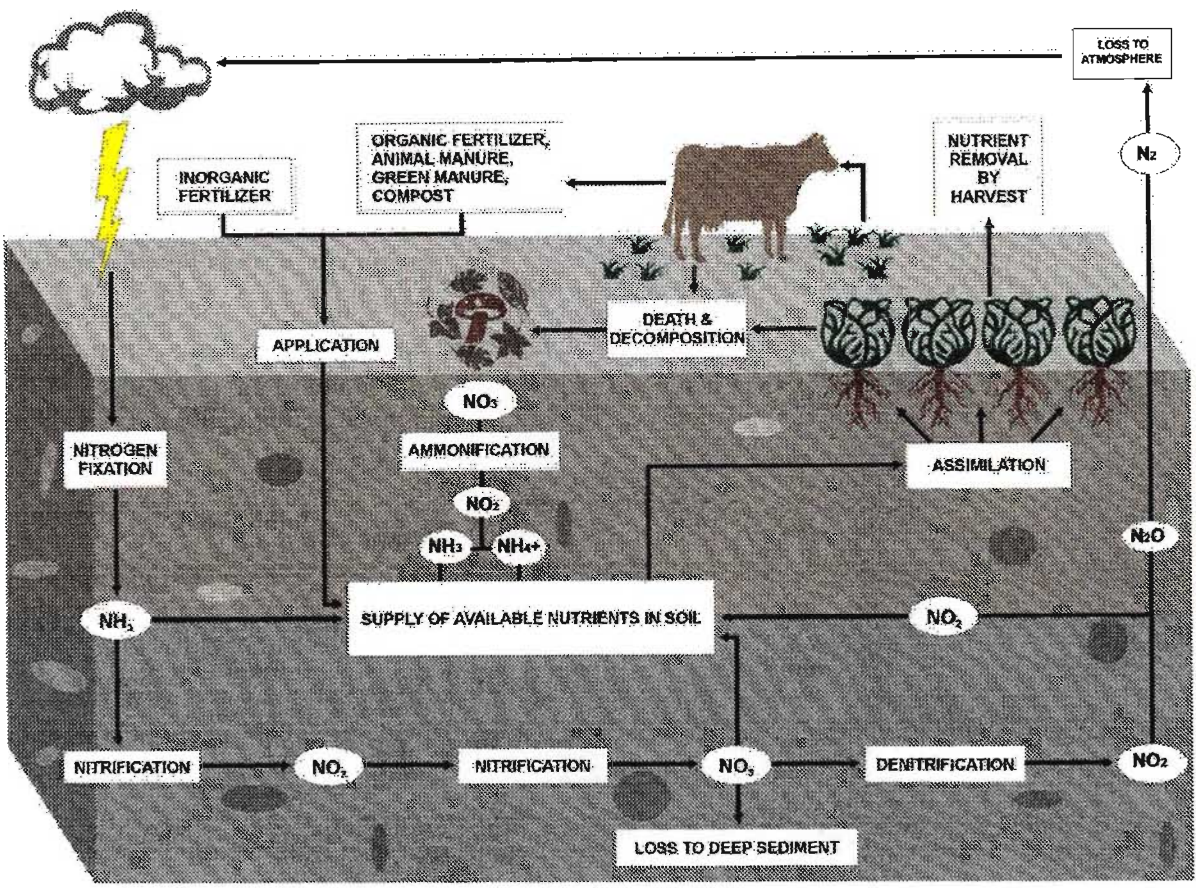


Figure 2.3: A diagrammatical representation of the nitrogen cycle

Animal and plant decay, and animal excreta decomposition, requires that fungi and bacteria convert nitrate (NO_3) back to nitrite (NO_2) and then to ammonia (NH_3) and ammonium ions (NH_4^+) in a process called ammonification (Groffman 1997). Superimposed on the conversion of proteins to nitrates, is the process of denitrification. Facultative, aerobic bacteria replace dissolved oxygen in anaerobic conditions with nitrate as an electron acceptor in the process of respiration (Flynn *et al.* 1999). Nitrate is converted into (N_2) and nitrous oxide gas (N_2O) which is lost to the atmosphere as gaseous nitrogen.

There are numerous inorganic and organic nitrogen forms, which are inter-convertible under suitable circumstances (CEP 2000). Riparian areas are capable of retaining, utilising and removing nitrogen in both forms.

2.4.2 Sediment-bound nitrogen

Most nitrogen in surface runoff is present as organic nitrogen, which is associated with suspended solids or sediment. Sediment refers to silt and soil particles that enter streams, rivers, lakes and other water bodies from eroding land (Figure 2.4). An understanding of sediment is necessary if organic nitrogen reaching aquatic resources is to be reduced. Therefore the movement of sediment is important when considering surplus organic nitrogen and the consequences for aquatic resources.

An impact of agricultural production on South African freshwater quality is the accelerated rates of soil erosion beyond the geological rates expected under natural conditions (Murgatroyd 1979). Schoeman and Scotney (1987) rate 13% of South Africa as high in terms of vulnerability to erosion. Further to this estimate, 40% of the country (and 40% of those parts dominated by water erodible soils) coincide with areas of moderate to high rainfall erosivity potential (Schoeman & Scotney 1987).

Verster *et al.* (1998) report over three million hectares of South African land is rendered unproductive as a result of severe soil erosion. It is estimated that soil formation in South Africa varies between 0.25 and 0.38 tons per hectare per year (Verster *et al.* 1998). It is further estimated that the average soil loss in South Africa is about 2.5 tons per hectare per year (Schoeman & Scotney 1987), that is ten times the rate of soil formation. The average rate of soil loss however, does not indicate the extremes.

The country's natural potential for erosion has become exacerbated by certain agricultural practices, where land mismanagement such as overstocking and overgrazing sections of land has made the problem worse. Rooseboom (1978) stated that most cultivated lands in South Africa may experience soil losses of between 0.1 tons per ha and 10 tons per ha. Kieck (1986) studied the erosion rates in pineapple plantations near Bathurst in South Africa, and estimated soil losses at about 60 tons per hectare per year. Other reports indicate that as much as 120 tons of soil per hectare per year may be lost during certain types of agricultural production (Verster *et al.* 1998).

Eroded sediment is transported from agricultural catchments, via streams and rivers, to the sea. Rooseboom (1978) estimates that 100 to 150 million tons of sediment exits South Africa, Lesotho and Swaziland every year. Of this amount, approximately 40 million tons of sediment per year is lost from the Orange River catchment alone. Murgatroyd (1979) studied sediment loadings of the Tugela Basin in Northern KwaZulu-Natal between 1950 and 1959. It was reported that the mean annual export of suspended sediment from the Tugela river amounted to approximately 463 tons per km^2. The author estimated the annual geological rate of erosion and sediment transport in the Tugela Basin at about 16 tons per km^2, which indicates an increase of 28 times the geological rate. The Tugela Basin was also studied by Martin (1987) almost thirty years later. The sediment supply to the Tugela River was estimated at 15 times higher than those in the geological past (Martin 1987). The author suggests that these increases in sediment supply may be ascribed to human influences such as agricultural malpractices, as has been seen in places like the eastern United States and parts of Tanzania.

Prosser *et al.* (1999b) also attribute an increase in sediment delivery to certain agricultural practices, including:

- the clearing of natural vegetation for the purpose of intensive crop and grazing lands, which has dramatically increased erosion rates
- the application of nutrient-rich fertiliser, which can be leached or eroded into the water source
- the clearing of riparian vegetation, reducing the efficiency of its sediment-trapping function, and increasing the destabilisation of banks and channel erosion
- the use of floodplains for agricultural production, unlimited stock access to streams, and the continuation of cropping up to the stream banks, bringing bare soil and nutrients into close proximity with the water source.

Eroded sediment commonly carries with it attached pollutants such as organic nitrogen and phosphorous from the fertilisers and pesticides used in agricultural production. Eroded sediment reaching water resources may contaminate human and livestock water supplies, smother breeding sites of aquatic life, and reduce the size and carrying capacities of streams and rivers (Section 2.4.2 and Section 2.5.2). Much organic nitrogen in eroded sediment would naturally be trapped and filtered by vegetation in riparian environments (Blackwell *et al.* 1999). Unfortunately however, a further impact of agricultural landuse is that much of the dense vegetation in these riparian areas has become denuded through encroaching agricultural practices (Rabie & Day 1998). Generally, the erosion-prone soils of South Africa, and the variable climate and scarce water resources make most of the country unfavourable for agriculture (Giliomee 1998). Increasingly more land is brought under agricultural activities to meet the demands of exponential population growth rates. The fertile environments of riparian soils make these areas more agriculturally favourable for cultivation. Veld-burning and animal overstocking are other agricultural malpractices impacting on South African riparian areas (Davies & Day 1986).

2.4.3 Nitrogen as a solute

Inorganic nitrogen is the sum of ammonium and nitrate (Altier *et al.* 1999). Nitrogen present as ammonium ions (NH_4^+) is a solute as it is formed when ammonia reacts with water. Nitrate is also a solute, and may enter aquatic systems in surface run-off, especially if heavy rain follows fertiliser applications (Gilliam *et al.* 1997). But more commonly nitrate enters riparian areas and aquatic resources in subsurface flow (Gilliam *et al.* 1997) (Figure 2.4).

2.5 CONSEQUENCES OF NITROGEN POLLUTION

2.5.1 Eutrophication

Nutrient enrichment by inputs of both phosphorous and nitrogen to lakes, rivers, estuaries and coastal oceans is termed eutrophication. The natural accumulation of nutrients is part of the ageing process of the water bodies on a geological timescale (O'Keeffe *et al.* 1998). This natural process may become accelerated by human activities, and as a result has become a common impairment of most surface waters (Figure 2.5). Impairment refers to the area of surface water no longer suited to its designated purpose.

Both marine and freshwater eutrophication can have negative effects on aquatic and terrestrial ecosystems, and can be detrimental to the health of humans. High inputs of phosphorous and nitrogen can boost the growth of aquatic weeds, increase the biomass of phytoplankton; benthic and epiphytic algae, and promote blooms of gelatinous zooplankton. Increased growth of the algae and aquatic weeds may interfere with activities of water utilisation such as consumption, fishing, recreation, agriculture and industry. Water quality problems related to blooms of cyanobacteria (blue-green algae) include unpalatability of drinking water, foul odours, decreased water transparency, and interference with water treatment processes (Carpenter 1998). During the decay of cyanobacteria, water soluble neuro- and hepatotoxins are released. This blue-green algae can pose serious health threats to humans if ingested, and similarly may cause death to livestock. Decomposition or senescence of other aquatic weeds and algae reduce the oxygen content of the system, causing anoxia and death of fish and other aquatic species. Highly toxic, volatile chemicals produced by some algae can cause long-term neurological damage to humans (Burkholder *et al.* 1992, as cited in Carpenter 1998). Excessive algal blooms in marine systems such as red and brown tides can be particularly destructive. Aquiculture and shell fisheries often suffer, as these algal blooms can cause shellfish poisoning in humans, mortalities in marine life, death of coral reefs and dependant communities, reductions in harvestable fish species, and oxygen depletion.

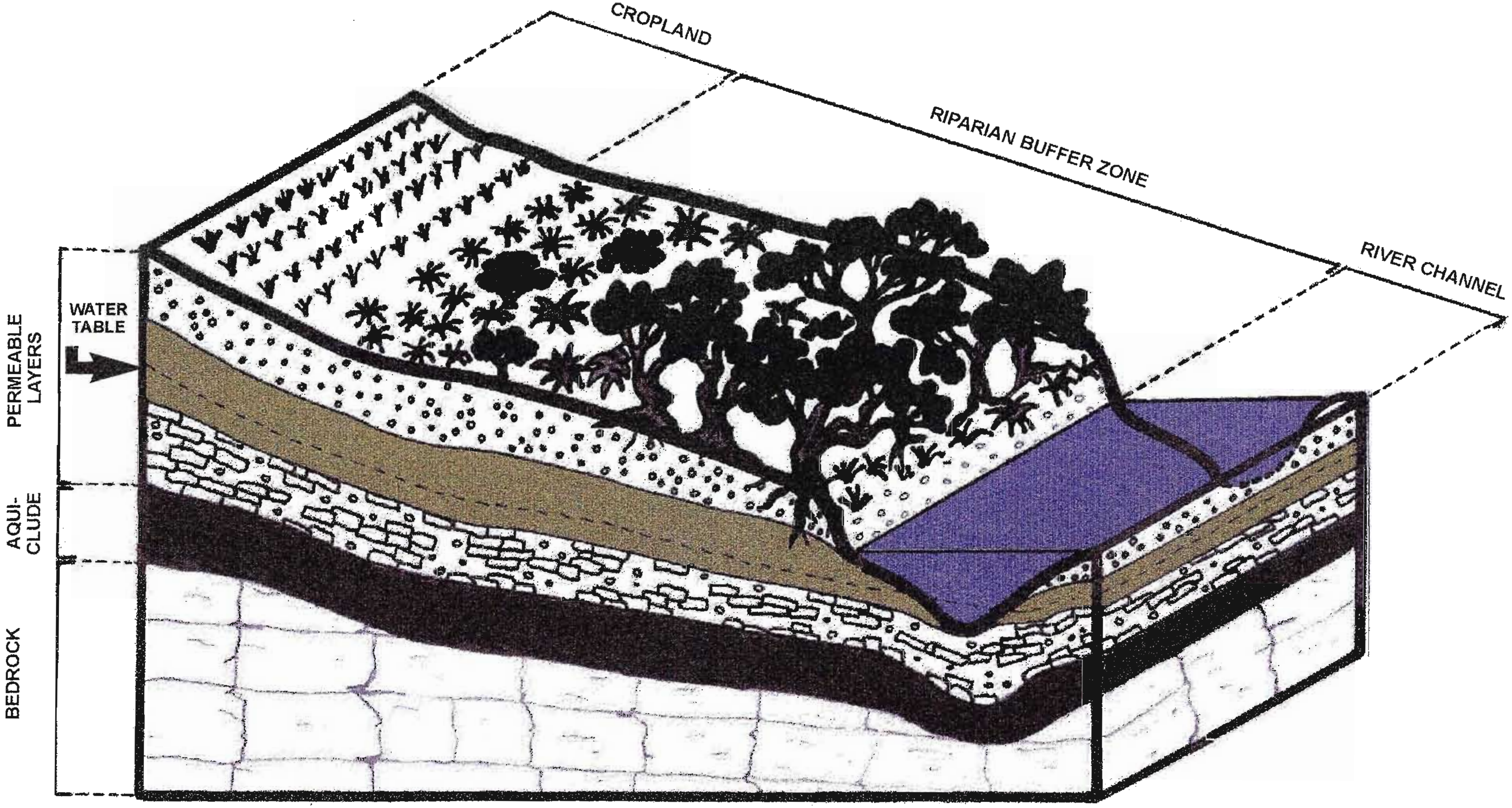


Figure 2.4a: Typical section through landscape showing cropland and riparian vegetation above a river channel, in relation to nitrogen cycling in Figure 2.4b.

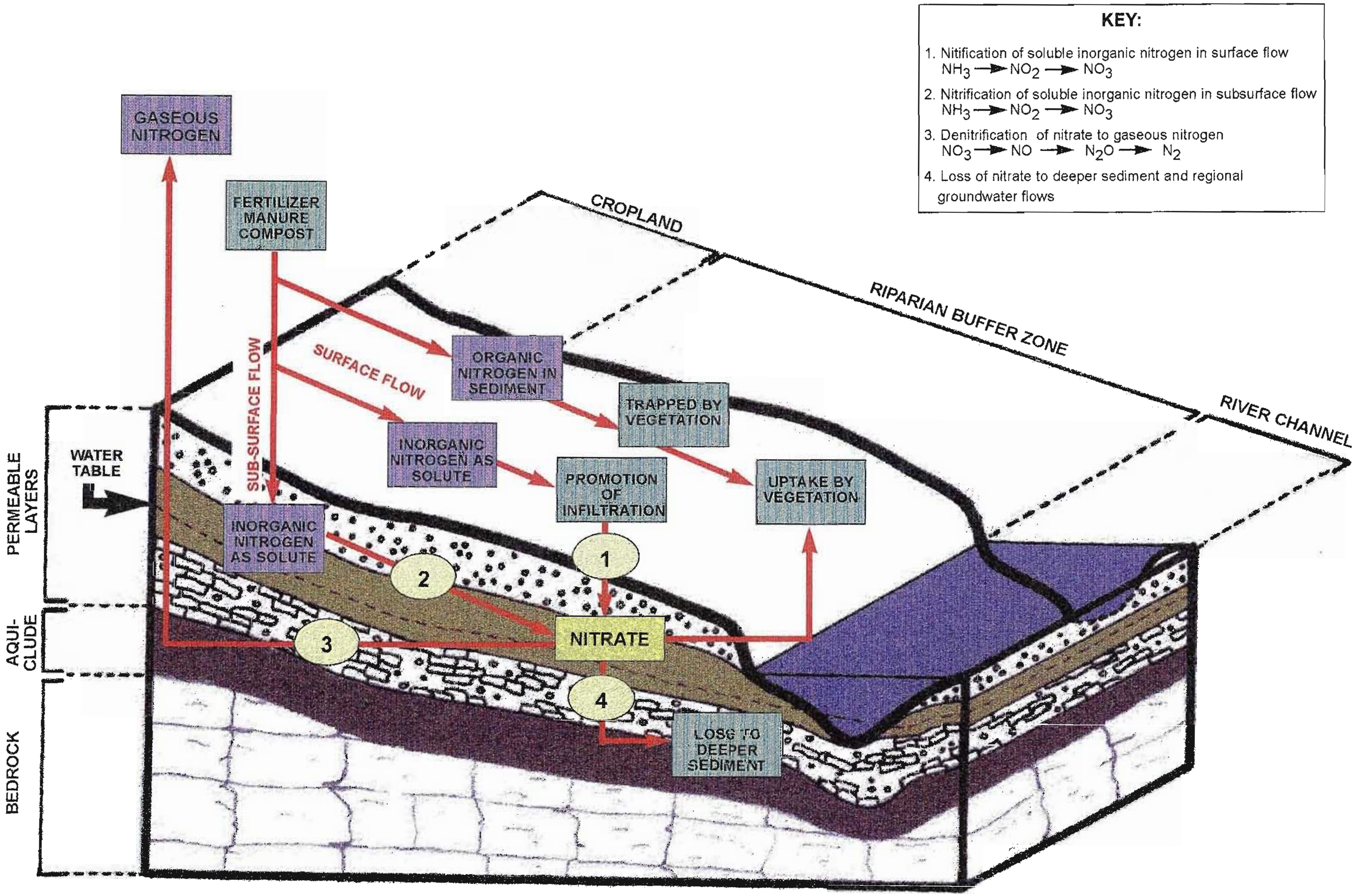


Figure 2.4b: Nitrogen entry and cycling processes within a riparian buffer zone

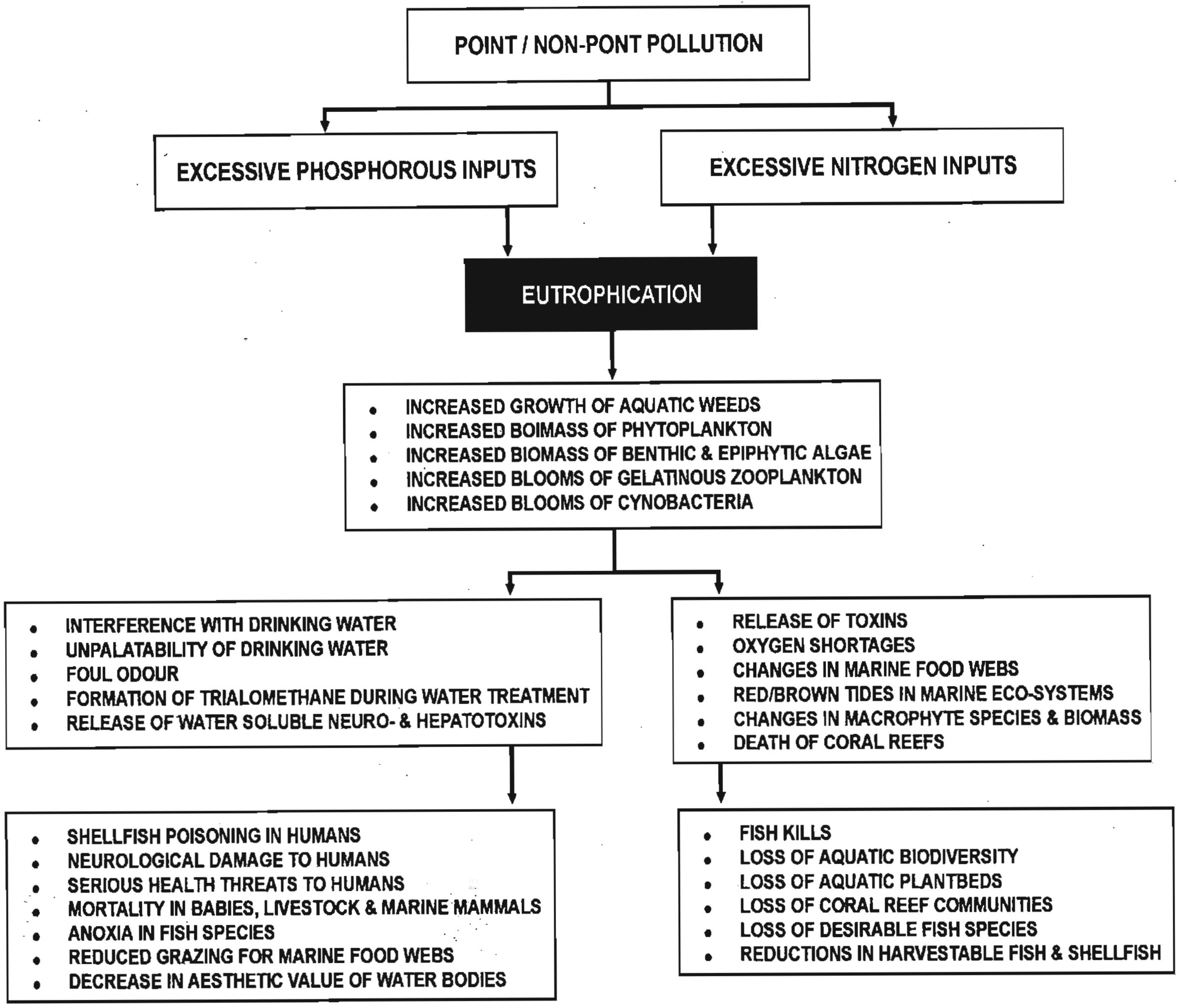


Figure 2.5: Problems associated with eutrophication of water bodies (after Dallas & Day 1993; Carpenter 1998)

The Hartbeespoort Dam in South Africa is an example of how an impounded water body may be affected by accumulating nutrients. The dam was described as oligotrophic when it was first constructed in 1928, but within thirty years had become eutrophic, and by 1985 was described by the National Institute for Water Research as hypertrophic (O'Keeffe *et al.* 1998). The eutrophied condition of the dam is of concern as the impoundment serves as source for irrigation and potable water supply. The dam experiences excessive growth of the toxic alga *Microcystis aeruginosa* and the water hyacinth *Eichhornia crassipes.* The sediment in the dam has shown considerable levels of heavy metal enrichment, and irrigation schemes may be affected by the concentrations of specific mineral constituents (Toerien & Walmsley 1978). Other South African water bodies affected by excessive eutrophication include the Msunduze River, which flows through Pietermaritzburg in KwaZulu-Natal; the Inanda Dam in Inanda, KwaZulu-Natal; the Black River which flows through Cape Town in the Western Cape; and the Buffalo River below Zwelitsha and King William's Town in the Eastern Cape (O'Keeffe *et al.* 1998).

2.5.2 Direct health effects

Nitrogen presents itself in many forms in both polluted and natural aquatic systems. Forms of nitrogen most commonly measured in water tests include ammonia (NH_4^+), albuminoid ammonia, organic nitrogen, nitrites (NO_2^-) and nitrates (NO_3^-) (Dallas & Day 1993).

i) Ammonia

Ammonia is a common pollutant associated with sewage and industrial effluent and found either in the free, un-ionized form (NH_3), or as ammonium ions (NH_4^+). Toxicity in ammonia is directly related to the concentration of the un-ionized form, as ammonium ions have little or no toxicity (Dallas & Day 1993). Ammonium in surface or ground water is generally derived from the decomposition of nitrogenous organic matter. Ammonium hydroxide results from the reaction of ammonia gas (readily soluble in water) and water (Sprent 1987). The ammonium hydroxide then dissociates into ammonium and hydroxyl ions, which tend to raise the pH value of the water (Dallas & Day 1993).

The amount of undissociated ammonium hydroxide in a solution is directly related to the toxicity of ammonia and ammonium salts to aquatic animals. The ammonium ion dominates at a low to medium pH, but as pH increases, ammonia, which is considerably more toxic, is formed (Dallas & Day 1993). The effects of toxic ammonia on the aquatic environment are many, for example:

- un-ionized ammonia affects the respiratory systems of many animals, either through cellular metabolism inhibition, or by decreasing the oxygen permeability of the cell membrane (Gammetre & Frutiger 1990 as cited in Dallas & Day 1993).
- acute toxicity to fish may cause loss of equilibrium, hyper-excitability, increased breathing rates, cardiac output and oxygen intake, or even convulsions coma and death (Hart *et al.* 1992 as cited in Dallas & Day 1993).
- chronic effects include a reduction in hatching success, growth rates, and morphological development, and pathological changes in tissues in gills, liver and kidneys (Hart *et al.* 1992 as cited in Dallas & Day 1993).

ii) Nitrite

Nitrite is a naturally occurring anion in fresh and saline waters. Human activities that increase nitrite concentrations in aquatic environments include industrial production of metals, dyes and celluloids, sewage effluents and certain types of aquiculture. Nitrite, the intermediate in the conversion of ammonia to nitrate, is toxic in certain concentrations (Dallas & Day 1993). Toxicity resulting from nitrite promotes the formation of methaemoglobin, which lacks the capacity to bind with oxygen, resulting in anoxia during high oxygen demand (Dallas & Day 1993). Fish during high levels of activity, have been found to die of anoxia, as their demand of oxygen during these times is higher (Tamm 1991).

iii) Nitrate

Nitrate is the end product of the aerobic stabilization of organic nitrogen. Nitrate may enter aquatic systems through agricultural run-off and fertilizer contamination. Photosynthetic action constantly converts nitrate to organic nitrogen in plant cells, and nitrate is therefore seldom found in natural surface waters. Nitrate is however, often found in high concentrations in ground water (Dallas & Day 1993). Nitrate pollution is considered a direct health threat to aquatic environments, mammals and humans (Carpenter *et al.* 1998). High concentrations of nitrate in water is toxic. Infants under the age of six months are most sensitive to nitrate pollution, as the microorganisms found in their stomachs reduce nitrate to nitrite. Nitrite causes the conversion of haemoglobin to methaemoglobin, which interferes with the oxygen carrying ability of the blood (Carpenter *et al.* 1998). The result in humans is termed "Blue-baby syndrome" or methemoglobinemea, and can lead to brain damage or death by suffocation in infants (Martin *et al.* 1999). Livestock experience a similar condition called asphyxiation, where bacteria in their digestive tracts reduce nitrate to nitrite, rendering the haemoglobin cells in the blood unable to carry oxygen. Other consequences resulting from the ingestion of nitrate may be reactions in the stomach to form carcinogenic, mutagenic and or teratogenic nitrosamins in the body (Starr & Gillham 1993 as cited in Martin *et al.* 1999), although this is not conclusively proven.

2.6 RIPARIAN BUFFER ZONES: A POSSIBLE SOLUTION

Non-point source pollution control requires a systematic approach, where source load reductions, best land management practices, and restoration or creation of pollutant sinks is achieved. Examples of such sinks include natural wetland systems, constructed wetland systems, and riparian buffer zones (Blackwell *et al.* 1999). The restoration of riparian vegetation or the design of riparian buffer zones to attenuate sediment-bound pollutants and solute pollutants from entering waterways is becoming more recognised as a solution to some of the environmental consequences of agricultural production. The creation of buffer zones between the aquatic resources and the terrestrial landuses may reduce the adverse impacts of the adjacent developments on the aquatic functions and values of systems (Castelle *et al.* 1992).

2.6.1 Riparian buffer zones

It is common practice to create buffer zones between adjacent landuses that are significantly different, or where the potential for conflict is serious (Karrsies & Prosser 1999). According to Palone and Todd (1997), the magnitude or density of the activity, or the severity of the impact generally results in the proportional increase of the buffer width to contain the negative impact. Buffer zones are common between airports or industrial activities and residential landuses, or between transport networks and houses.

There are significant differences between aquatic environments and adjacent developed or disturbed lands. These differences, coupled with the increasing degradation and overutilisation of natural riparian areas (Barling & Moore 1994), necessitate the creation of riparian buffer zones. Dosskey *et al.* (1999) define a riparian buffer as land adjacent to streams, lakes and wetlands that is managed for perennial vegetation (grass, shrubs and/or trees) to enhance and protect aquatic resources from the adverse impacts of agricultural practices.

A riparian buffer zone provides an important link between the terrestrial upland ecosystem and the aquatic stream ecosystem (Osborne & Kovacic 1993). The term 'riparian buffer zone' refers to vegetation placed between the hill slopes and streams (or small drainage lines) for the stabilization of soil; filtering and reduction of sediment input and soil-attached pollutants to waterways from up slope agricultural areas; habitat creation for wetland-associated species; aesthetic value and noise pollution reduction; and the overall reduction of human disturbance to the aquatic system (Castelle *et al.* 1992; Bren 1993).

A riparian buffer zone can be useful in maintaining a productive, profitable and responsible farming operation (Table 2.3). A buffer zone can be installed at field edges or within fields, alongside streams, lakes and rivers. It is noted however, that good management of a riparian buffer zone is an insufficient substitute for good land management practices, but is increasingly being recognised as a necessary supplement to the overall farming system. By incorporating riparian management into farming practices, landholders can achieve long-term economic and environmental sustainability of their properties and water catchments as a whole. Riparian buffer zones should be considered by farmers utilising cropland, grazing land, livestock enclosures and pasture, especially those whose properties directly adjoin a water course. While the design and vegetation type of these buffers may differ according to the context in which they are needed, their buffering functions of surface and sub-surface water quality protection are similar.

Table 2.3: The diversity of riparian functions (after Karssies & Prosser 1999; Klapproth 1999; LWRRDC 2000)

Functions of a riparian buffer zone
• the control of stream temperature and light
• the control of aquatic plant growth
• the maintenance of in-stream habitat
• the provision of shade, shelter and food for aquatic/riparian life
• the reduction of sediment delivery from agricultural runoff
• the stabilisation of stream banks, and protection against erosion
• the modification of water quality through the filtering of nutrients, pesticides and pollutants
• the provision of aesthetic, recreational and economic value to the landscape

2.6.2 Water quality functions

There are several ways in which a riparian buffer zone can influence rivers, streams and water bodies (Table 2.3). This mini-dissertation is however, particularly concerned with a buffer zone's ability to attenuate sediment-bound and soluble nitrogen thereby improving the water quality of the catchment system. The water quality amelioration function of riparian buffer zones will be discussed in greater detail.

The most important functions of a riparian buffer zone in terms of water quality preservation is the reduction of sediment and soluble nutrients and pollutants. The reduction of sediment and nutrient delivery is three-fold (Karrsies & Prosser 1999):

- Erosion is reduced by good vegetation cover
- Infiltration is promoted by good vegetation cover
- Sediment and attached nutrients are trapped by good vegetation cover.

i) Prevention of accelerated erosion

Valuable agricultural land can be lost to eroding or collapsing banks, and the increased sediment to streams may become problematic (Lynch *et al.* 1985). In some instances, the riparian environment itself may become susceptible to erosion, especially if it is over-utilised or is of particularly steep gradient. The riparian buffer zone prevents the erosion of the soil both within and above it's environment, specifically through the prevention of headward gully erosion.

The contribution a riparian buffer zone can make towards erosion control is a result of the vegetation structure and composition (Karssies & Prosser 1999). Erosion can be prevented by vegetation cover, as the plants bind the soil with their root systems thereby stabilising steep banks (Martin *et al.* 1999), and they improve soil aggregate stability through the production of leaf-litter and the promotion of infiltration. A study in Zimbabwe (Whitlow 1983 as cited in O'Keefe *et al.* 1998) indicates the role vegetation can play in erosion control. Erosion rates were tested on Sandveld soils for different land-uses. Bare plots lost a mean annual amount of 4 320 kg of soil per hectare, but protected plots lost only 200kg of soil per hectare per year, almost 22 times less than the bare plots.

Agricultural practices on floodplains are common because of the fertile nature of the soil (Verster *et al.* 1998). These practices can be particularly detrimental in terms of erosion and degraded water quality (Karssies & Prosser 1999). Many agricultural practices in floodplains remove the natural vegetation, thereby increasing erosion and sedimentation rates, and reducing channel capacity. Reduced channel width increases velocity of flow. The vital role of the remaining natural vegetation in reducing the flood-flow velocity is jeopardised. Flood flows are often of such significant depth and volume, that riparian buffer zones are insufficiently capable of preventing erosion. However, healthy riparian vegetation is able to retain some overland flow, releasing it at a slower rate into the system, and is able to protect the soil from bank scour.

ii) Promotion of infiltration

Infiltration is one of the most significant removal mechanisms affecting water quality by buffer zone function (Dillaha & Inamdar 1997). The rate of infiltration is generally determined by the soil properties, vegetation type, climate, antecedent moisture conditions, and the amount and velocity of surface runoff (Karssies & Prosser 1999). Riparian environments are usually wetter than surrounding areas, and remain close to saturation point for long periods of time. It is usually assumed that riparian buffer zones are actually a source of run-off. Yet the dense vegetation of a riparian area is able to trap water in shallow pools, allowing it to infiltrate over time. The root and stem structures, and litter-fall reduce the velocity of water flow, also promoting infiltration. A thick litter layer improves the soil structure increasing resistance to erosion, and deep roots of riparian vegetation may provide infiltration channels into the soil (O'Keeffe *et al.* 1998). Finer sediment particles in suspension are deposited in the soil profile as infiltrated water moves through the soil mantle (Dillaha & Inamdar 1997).

Infiltration is important in that it is the main trapping mechanism for filtering solutes such as nitrates (Prosser *et al.* 1999c). Water needs to be stored in the soil for long periods of time so that plants are able to absorb it, and solutes may undergo microbial processes such as denitrification. Infiltration in riparian environments is promoted where there are small volumes of run-off, particularly in the form of sheet flow. The width of the riparian buffer zone plays an important role in the extent of infiltration, and it is generally found that the greater the width, the more effective the riparian land in the absorption of water. The soils of riparian areas are usually moist and organic rich, and this coupled with the vigorous plant growth in the riparian environment promotes the absorption and transformation of nutrients and other soil-attached pollutants (Prosser *et al.* 1999a).

iii) Filtering of sediment and nutrients

Sediment loading and deposition constitutes one of the most serious water quality problems throughout the world (Osborne & Kovacic 1993). The semi-arid regions and steep gradients of the rivers; the frequency of flooding; and the history of land mismanagement in many parts of South Africa promotes large sediment loads in South African rivers (Verster *et al.* 1998). In many agricultural catchments, the supply of sediment and nutrients to streams and rivers has increased. The clearing of catchments, soil disturbances, changes in vegetation cover, and over utilisation of riparian land environments have led to a substantial increase in the amount of sediment (sand, gravel, silt and clay) and attached pollutants entering the streams and rivers (Dallas & Day 1993).

Sedimentation of streams can have a pronounced affect on water quality and stream organisms (Gilliam 1994), such as the suffocation of fish eggs and aquatic larvae, the clogging and abrasion of fish gills, and the modification of reproductive and consumption behaviours of aquatic organisms (Karssies & Prosser 1999; Klapproth 1999). Sedimentation of a water body, reduces channel capacity, which increases the probability of flooding, promotes weed invasion, and interferes with recreational activities. Sediment may carry with it pollutants and excessive nutrients from agricultural run-off, which cause many water quality problems (problems relevant to this mini-dissertation are discussed in greater detail in Section 2.5). Increased sediment and nutrient delivery to a river system can result in high contaminant levels, which degrade the drinking water quality and the aquatic habitat (Dillaha *et al.* 1989).
These contaminants may be:

- faecal bacteria and other microbes in animal waste, capable of causing disease
- nitrates and pesticides toxic to human, animal and aquatic life
- phosphates which promote algae blooms, these blooms suffocate aquatic animals and organisms.

Riparian buffer zones are able to trap sediment as both the vegetation and generally lower gradient of these areas (Chaubey *et al.* 1995; Prosser *et al.* 1999b) reduces the velocity of flow and increases the surface roughness or flow resistance (Osborne & Kovacic 1993; Gilliam 1994). The reduction in flow velocity may cause the incoming sediment load to exceed the carrying capacity of the water, where the sediment will drop out of suspension (Martin *et al.* 1999). Particulate waste and sediment-attached contaminants are deposited with the sediment. The reduction in flow velocity within the riparian buffer zone is transmitted back to the area up slope. This gives rise to an area termed backwater (Karssies & Prosser 1999) where deposition in the deep, slow-flowing water of the riparian area progresses backwards towards the upland areas of the riparian buffer zone. The storage capacity is constantly renewed by vegetation germination and regrowth on and through the trapped sediment. The improved infiltration of surface run-off and the vigorous growth of riparian vegetation promote the uptake and transformation of soluble contaminants by plants and soil microbes (Dosskey *et al.* 1999). Soluble contaminants may be similarly removed from shallow sub-surface flow in contact with the riparian root systems (Faafeng & Roseth 1993; Peterjohn & Correll 1986).

2.6.3 Types of riparian buffer zones

There are many types of riparian buffer zones. While these zones may be referred to by different names in different parts of the world, their functions remain much the same globally. Certain vegetation types will prove more efficient at individual riparian functions than others, and the combination of vegetation type requires careful consideration of the desired overall role of the riparian area. It is therefore necessary to identify the desired purposes of the riparian buffer zone before considering the vegetation type and buffer zone design. Three types of riparian buffer zones are to be considered in greater detail, namely Riparian Forest Buffer Systems (FBS); Grassed Filter Strips (GFS); and Mixed Buffer Systems (MBS).

i) Riparian forest buffer systems

A Riparian Forest Buffer System (FBS) is a stream-side ecosystem (area of trees and shrubs located adjacent to streams, lakes, ponds and wetlands), managed for the enhancement of water quality and the protection of the stream environment through the control of non-point source pollution (Altier *et al.* 1999). The University of South Carolina, USA encourages a FBS of diverse plant types; plant heights; root depths; stem densities; canopy cover; and vegetation of varying ages, as this will be most efficient at a range of riparian functions (CEP 2000). A FBS of sufficient width is useful in the interception of sediment, nutrients, pesticides, and other materials in surface and sub-surface runoff. A good revegetation programme targeting riparian areas will promote the planting of indigenous trees and shrubs, as these are considered more effective in water quality amelioration than exotic species (Bren *et al.*1997). A FBS should be established concurrently with other practices as part of a conservation management system.

It is important to control water flows and erosion up slope from the FBS if the area is to maintain proper functioning. Periodic harvesting of certain trees within a FBS will also help maintain plant health and buffer function (NRCSb 1997).

ii) Riparian grass filter strips

Grass Filter Strips (GFS) are areas of grass planted between the hill slope and streams (or small drainage lines) to reduce the input of sediment and soil-attached pollutants to waterways from up slope agricultural areas. The effectiveness of GFS in trapping sediment will vary according to water flow and inflow rates of nutrients and sediment, type of grass used, and buffer strip configuration. A revegetation programme should consider the type of sediment-attached pollutants it wishes to target, and then utilise the most effective GFS for that particular function. If the GFS is to perform the dual function of buffer zone and crop, it is important to plan rotation practices, nutrient and pest management, crop residue management, and other cropland practices. GFS are most effective in providing conservation benefits when used in combination with other agronomic or structural practices (NRCS 1997a).

iii) Riparian mixed buffer systems

Many studies are currently investigating the use of a Mixed Buffer System (MBS) as a stream-side management practice (Lowrance *et al.* 2000a; Lowrance *et al.* 2000b; Pinay & Burt 2001). A MBS requires that the riparian area be divided into three lateral zones of different widths, functions and vegetation types (CEP 1999). Each of the zones will be managed according to its specific riparian function and will therefore have its own management practices. Typically, most management practices of MBS require increased levels of management away from the stream (Lowrance *et al.* 2000a) (Table 2.4). The three zones follow parallel to the stream: the streamside zone; the middle core zone; and the outer core zone (Table 2.4).

Table 2.4: Three-zone approach to riparian management: Riparian Mixed Buffer System

ZONE	DESCRIPTION
THE STREAMSIDE ZONE (Undisturbed forest)	Usually an area of undisturbed mature riparian forest for the protection of the physical and ecological integrity of the stream itself (CEP 1999). The zone provides important habitat for both terrestrial and aquatic species, controls stream temperature and stabilizes stream banks (Lowrance *et al.* 2000a). The undisturbed nature of the forest requires that landuse be restricted within this zone, to services such as footpaths and utility roadway crossings.
THE MIDDLE CORE ZONE (Managed forest)	Usually an area of managed woody vegetation (Lowrance *et al.* 2000a) for trapping and filtering sediment; nutrients and attached pollutants from upland runoff. The stream order, the extent of the 100 year floodplain, the gradient of the stream banks and riparian area, the proximity of adjacent wetlands, and the intensity of the upland landuse should all be considered when defining the boundaries of the middle core zone. The zone may be managed as a natural mature riparian forest, but some clearing and planting may be permitted (CEP 1999). Recreational activities such as bike trails may also be allowed in this zone.
THE OUTER CORE ZONE (Runoff control)	Usually an area planted as a herbaceous grassed filter strip for the dispersal of upland runoff; and the promotion of infiltration and sediment and nutrient deposition (Lowrance *et al.* 2000a). There are few landuse restrictions to this area, and its width will depend on the gradient, and intensity of upland runoff (CEP 1999).

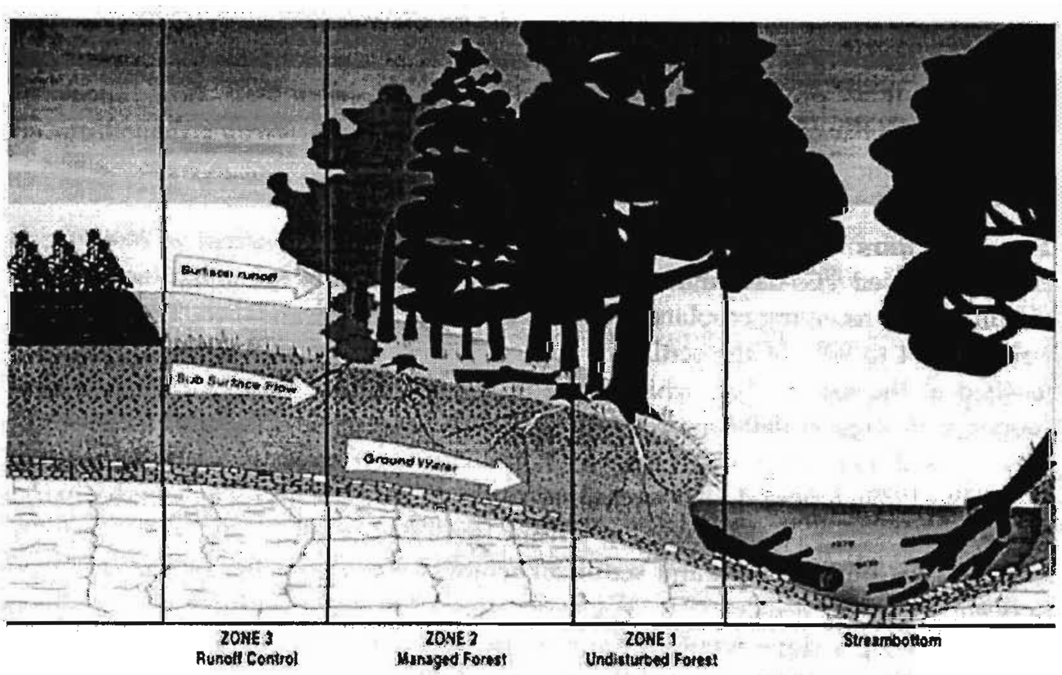


Figure 2.6: Diagram of mixed (three-zone) buffer system (Lowrance *et al.* 1995)

2.6.4 The recognition of riparian buffer systems

The Pacific Northwest, USA have policies and guidelines protecting riparian ecosystems (Cohen *et al.* 1987). These guidelines have been established to mitigate river ecosystem problems associated with agricultural production, residential encroachment and land mismanagement. The authors anticipate that new policies and regulations for the protection of riparian systems may have significant impacts on preservation of water quality and creation of habitat, with associated benefits to all water managers and users.

Klapproth (1999) investigated the lessons learned from several multi-agency stream restoration projects throughout North Carolina, USA. Riparian buffer zones were essential management practices of the projects, for achieving river channel stability and pollution prevention. The effectiveness of the riparian buffer zones was largely dependent on landowner cooperation, and the input of many experts.

The Pennsylvania Department of Environmental Resources, Bureau of Forestry, USA developed a set of Best Management Practices to limit and control non-point source pollution from silvicultural activities (Lynch *et al.* 1985). The use of protective buffer strips was recommended along perennial and intermittent streams. Similarly the State of Environment Australia (Kapitzke 1999) recognise the establishment and management of riparian vegetation as important in the overall rehabilitation of Australian rivers. Establishment of riparian buffer zones is part of a tiered approach to resource management adopted by the State of the Environment Advisory Council. It is envisaged that these approaches will provide a sustainable water resource and ecology.

A building-block model for stream restoration has been devised in Sweden, based on international literature (Petersen *et al.* 1992). The model suggests the use of riparian buffer zones as a restoration measure, as a result of the present conditions of streams in the agricultural landscape. Riparian buffers as wetlands, riparian swamp forests or filter strips are recognised by the authors as important in river restoration, particularly in the reduction of nutrient delivery to groundwater, freshwater systems and the sea.

2.6.5 Riparian buffer zone efficiencies

The water quality buffering effects of vegetated riparian zones have been studied since the early 1970s (Correll 1997). The initial focus of the early studies was on the sediment buffering effects of riparian zones, but by the late 1970s the importance of streamside vegetation on stream environment and health was being realised. Many of the initial studies focussed on surface processes, but in the early 1980s three independent studies (Lowrance *et al.* 1983; Peterjohn & Correll 1984; Jacobs & Gilliam 1985) indicated that subsurface water quality could be improved by buffer systems. This observation was also supported by Gillliam *et al.* (1997). Correll (1997) identified at least 400 papers investigating the water quality buffering effects of vegetated riparian zones by 1997, and the rate of publication was about 30 to 35 papers per year.

Many reviews have indicated that high pollutant and nutrient removal efficiencies in buffer zones can be expected. Many of the studies world wide regarding nutrient attenuation suggest that high nitrogen attenuation can be achieved in riparian buffer zones. Both organic and inorganic nitrogen from agricultural surface and subsurface flows can be attenuated by riparian buffer zones. Design and management of these riparian buffers specifically for the function of nitrogen attenuation is possible through an understanding of the means by which these buffer zones retain or remove nitrogen. The design of riparian buffer zones requires recommendations regarding how to decide on the most efficient buffer size without compromising valuable farming land. One of the aims of this research is to better understand the processes of nitrogen attenuation in riparian buffer zones, such that making recommendations regarding buffer size for this function is made easier.

There are many variations in recommendations regarding the size of buffer necessary for achieving the best possible buffer function. Figure 2.7 is an indication of these variations. Three reviews by Castelle (1994), Palone and Todd (1997), and Dosskey have suggested the minimum to maximum width required for the desired buffer functions.

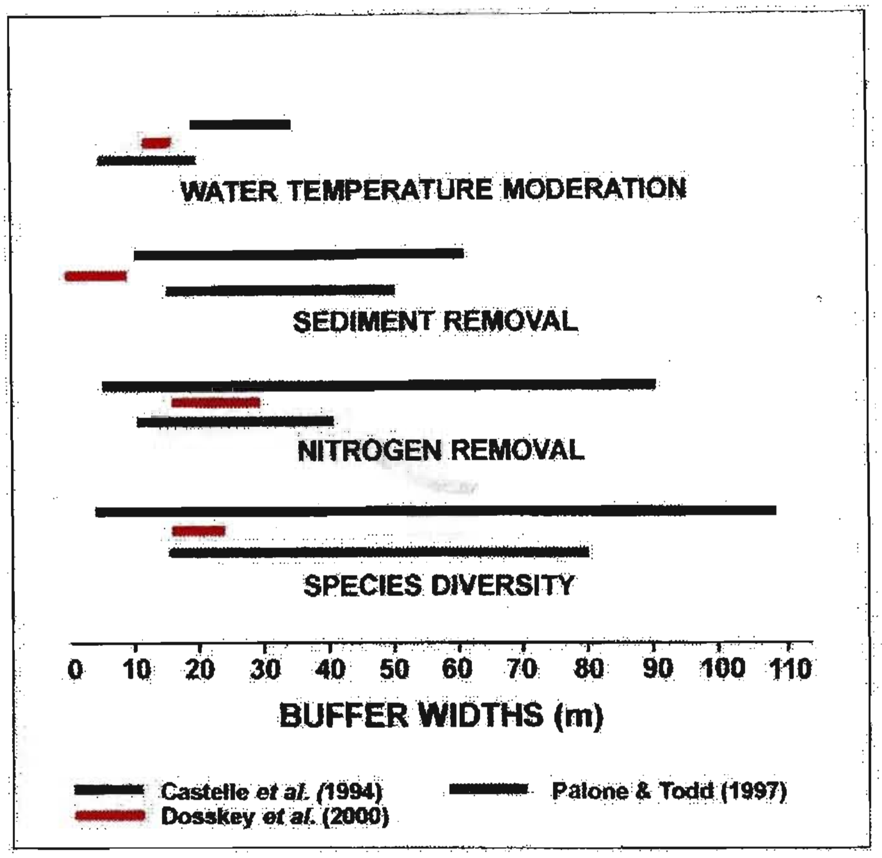


Figure 2.7: Recommended buffer sizes for specific riparian functions (after Castelle *et al.* 1994; Palone & Todd 1997; Dosskey *et al.* 2000)

The diversity in existing recommendations regarding riparian buffer zone sizing for nitrogen attenuation, and more specifically nitrate attenuation, makes implementing these zones a difficult process. The reviews suggest that a buffer width anywhere from five metres to ninety metres is necessary for efficient nitrate attenuation. A better understanding of the characteristics in riparian buffer zones responsible for enhancing nitrogen attenuation may assist in narrowing the range of minimum buffer widths recommended. It can be seen that the sizes suggested for maximum nitrogen removal efficiencies are generally larger than those necessary for sediment retention. The size of a riparian buffer zone for soluble nitrogen attenuation will therefore also suit the function of sediment-bound nitrogen attenuation. For this reason sediment retention will no longer be considered in this mini-dissertation, as any width suitable for trapping solute nitrogen will be sufficient for trapping sediment-bound nitrogen.

3. UNDERSTANDING THE RIPARIAN BUFFER ZONE FUNCTION OF NITRATE ATTENUATION

3.1 INTRODUCTION

Soluble nitrogen in the form of nitrate has received the most attention in studies involving nutrient attenuation by riparian buffer zones (RBZ's) (Gilliam *et al.* 1997). This may be attributed to its potential toxicity to both humans and animals at concentrations greater than 10mg/l in potable water supplies (CEP 2000). Hill (1996) attributes the recent interest in nitrate to the previous lack of scientific information about riparian buffer nitrate attenuation functions. From the recent studies, there is general agreement that riparian buffer zones are the most important factor controlling entry of non-point source nitrate, especially as these zones are able to sequester nitrate from both surface and subsurface flows (Peterjohn & Correll 1984; Jacobs & Gilliam 1985; Lowrance *et al.* 1984c, 1997; Osborne & Kovacic 1993; Hill 1996; Lowrance 1997).

The mini-dissertation will therefore focus on nitrate as an example of soluble nitrogen, in the understanding of riparian buffer zone nitrogen attenuation processes. This chapter will explore the mechanisms of nitrate attenuation in riparian buffer zones. Those factors influencing these mechanisms will be discussed at an organism, and at a landscape level. Through an understanding of the influential factors controlling the mechanisms of nitrate attenuation and removal, it is possible to improve recommendations regarding buffer assessment, design and management.

3.2 MECHANISMS OF NITRATE ATTENUATION IN RBZ's

There is relative uncertainty regarding the key mechanism responsible for the removal or immobilisation of nitrate in riparian land environments. Evidence (Lowrance *et al.* 1984a, 1984b; Peterjohn & Correll 1984; Cooper 1990; Jordon *et al.* 1993) suggests that nitrate depletion in riparian ecosystems is not only the result of its conversion to other soluble forms of nitrogen, as loss of nitrate recorded in riparian areas is seldom associated with increased amounts of NH^{+}_{4} and organic N in subsurface water (Hill 1996). Riparian environments are therefore important sinks for nitrate removal and retention. Denitrification, assimilation and retention by vegetation, utilisation by soil microfauna, nitrification, and mineralisation or transformation to ammonium and organic nitrogen, are all processes noted to retain, remove or immobilise nitrate (Correll 1997).

Two key processes that immobilise and/or remove nitrogen from surface and subsurface flows are generally recognised, either individually or together, in the available literature on the water quality amelioration functions of riparian land environments. These two mechanisms of nitrogen removal are to be discussed in greater detail, and supported by various studies and findings.

- The retention or immobilisation of nitrogen by plants and organisms (vegetative uptake) as recognised by Hemond and Benoit (1988); Peterjohn and Correll (1984); Lowrance *et al.* (1984b); Fail *et al.* (1986.
- The removal of nitrogen into the atmosphere through the process of denitrification is discussed in studies by Jacobs and Gilliam (1985); Cooper (1990); Ambus and Lowrance (1991); Haycock and Pinay (1993).

3.2.1 Vegetative uptake

Plants may be a temporary storage of nutrients, retaining the nutrients within the riparian zone for long periods of time. These nutrients will later be released by mineralisation. Results from 13 sites studied in the Nicolas Project (Pinay & Burt 2001), indicate that the residence time of nitrogen in a riparian landscape is significantly increased by vegetation processes. It was estimated that in some cases nitrogen was retained by vegetation from 1 to several years.

Plants require nitrogen as a macronutrient in mineral nutrition, but most green plants are unable to utilise element nitrogen. The complex molecules must first be converted into available forms through the action of soil organisms (certain prokaryotic organisms). Nitrogen fixation is the conversion of element nitrogen into organic ammonium by these organisms, and requires anaerobic conditions and energy in the form of ATP (Barbour *et al.* 1987). Once element nitrogen has been converted to available forms, it can be metabolised by plants, and used to meet the nitrogen demands of the cell.

Sodium nitrate; potassium nitrate; ammonium nitrate; and calcium nitrate are key available forms of nitrogen for green plants (Weier 1982). Plants utilise nitrates in the formation of structural units such as proteins, and other forms of nitrogen in organic compounds such as chlorophyll; nucleic acids, amino acids; alkaloids; and plant hormones (Weier 1982). The accumulation of nitrate by plants and microorganisms is not removed from the system but merely immobilised (Correll 1997). Nitrate can be stored or recycled by plants, accumulating in young tissues; seeds; and storage organs (Barbour *et al.* 1987), and thereby reducing the release of nitrate to the stream system.

Plants are only able to absorb substances in solution. Water and attached solubles flow continuously through the soil; the root; stem; and leaf of the plant, and into the atmosphere. Emergent vegetation utilises nitrate primarily available from the substrate, where rooted plants can assimilate nutrients from both the soil and water, and floating plants take up the nutrients directly from the water. As mineral elements are limited in mobility in the soil, plants have developed deep root systems with root hairs, which provide a large surface area to be in contact with a great mass of soil.

Water enters the root primarily through the root hairs of the plant (Figure 3.1). Nitrates in solution are absorbed through the cell wall and cytoplasmic membranes of the root hairs, as the soil solution is usually much less concentrated than the cell sap. Water (holding nitrate in solution) diffuses inward because the water potential in the plant is much lower than that of the soil (Weier *et al.* 1982). It follows a path of least resistance through cell walls and the cortex, to the endodermis where resistance increases. The casparian strip makes the cell wall impermeable, and the water is forced through the cell membrane. The passage of water and dissolved minerals from the endodermis into the vascular cylinder and primary xylem offers considerably less resistance. The water and solutes enter the leaf, where the vascular strands divide into fine segments so that almost every cell is in close proximity to the vascular tissue. From the vascular tissue, water travels through cell walls to the stomatal cavities, where it vaporises and travels out through the stomatal pores into the atmosphere (Barbour *et al.* 1987). Nutrients in solution are utilised by the plant in different ways, and therefore removed from the plant-water in different areas of the plant.

The uptake of nitrogen by plants as ammonium ion or nitrate ion increases as the nitrate concentration in the soil increases, but only to a point, where the rate then levels off. Translocation and distribution of nitrogen in the plant varies seasonally. Nitrate may be utilised in green shoots during growth seasons (spring and summer), and then translocated below ground for dormant seasons (winter) (Hemond & Benoit 1988). This is indicated in the change in ratio of live shoot nitrogen to live root nitrogen concentration. The ratio of shoot to root nitrogen varies from 3.0 to 1.0, changing with plant phenology. This ratio may cause the translocation of nitrate below-ground for dormant seasons before aboveground biomass dies back, so that living roots may utilise it (Barbour *et al.* 1987). The senescence, death and decomposition of the plants release the detained nitrate back into the soil, which may then be returned to solution through the leaching of nutrient-rich litter. Many nutrients released by plant senescence and decay may be recycled through microbial, physical or chemical attenuation mechanisms in the root zone of the riparian soils (Palone & Todd 1997). The re-released nitrate will only leave the system in a soluble form, or when plant litter is washed away.

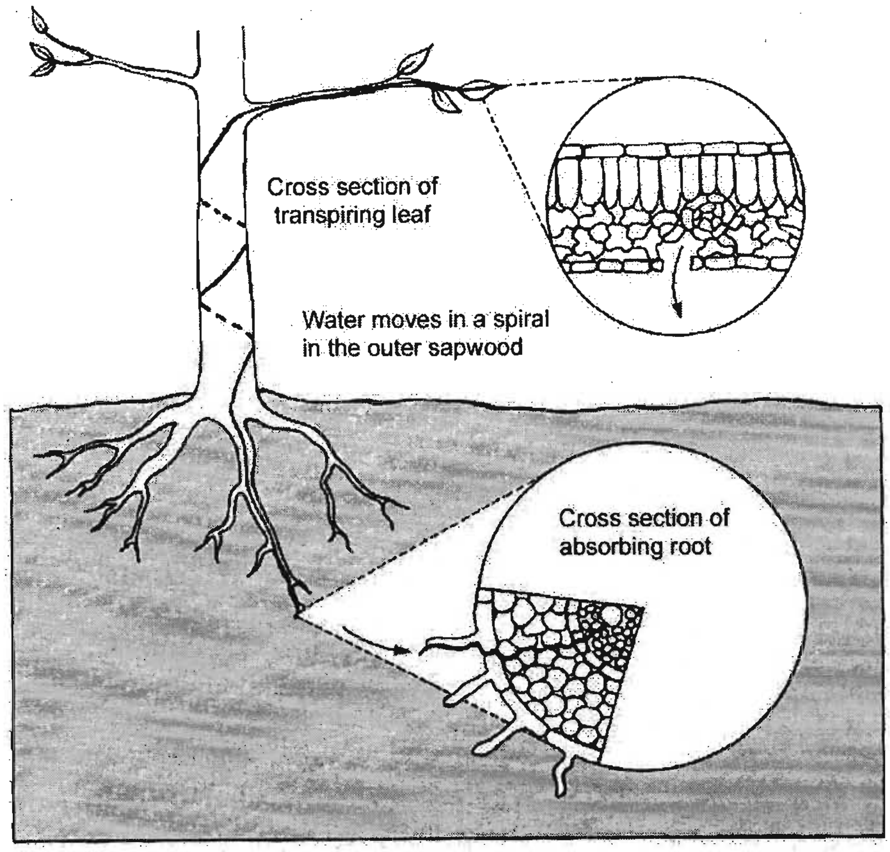


Figure 3.1: Diagrammatical representation of the pathway of water movement through a plant (Barbour *et al.* 1987)

Important characteristics of riparian vegetation for the determination of uptake potential include the structure, density and condition of the plants (Karssies & Prosser 1999). The uniform spacing of stems and greater vegetation density will be most effective at sediment trapping and retention. Soluble pollutants or nutrients are more likely to be taken up by vegetation if they are retained within the riparian environments for longer periods of time. The rate of nutrient cycling will depend on three things: the longevity of the plant parts; the refractory components of the litter; and the suitability of the environmental conditions for decomposition (Pinay & Burt 2001). Vegetation of good condition will improve the storage capacity and infiltration rate of riparian soils, thus increasing nitrate trapping via the infiltration mechanism (Prosser *et al.* 1999c). Senescent vegetation supplies organic matter and carbon to the soils, promoting the nitrate removal mechanism of denitrification.

Riparian areas usually have plant species which are tolerant of frequent flooding (Brinson 1988), as they are able to sequester nitrate in low-oxygen conditions by means of metabolic responses (Palone & Todd 1997). These species have morphological adaptations that facilitate the availability of oxygen and prevent root anoxia. In the event of prolonged flooding, flood-tolerant species are able to increase the thickness of their roots which increases porosity, allowing an internal downward diffusion of oxygen.

3.2.2 Denitrification

In addition to vegetative uptake, microbial processes in the soil attenuate pollutants in riparian land environments. Soil micro-organisms, like plants, take up and convert nutrients to less biologically available forms which are more readily stored in the soil (Klapproth 1999). Death and decomposition of the microbial cells re-release these immobilised nutrients into the soil (Palone & Todd 1997). These microorganisms not only immobilise nutrients in the aforementioned way, but also utilise and metabolise pesticides and other organic chemicals as an energy source. These organic chemicals are transformed to less toxic compounds during microbial processes (Klapproth 1999). Finally, the degradation of organic pollutants, and many chemical reduction reactions in the soil, including denitrification of nitrates, are the result of soil microbial activities.

Denitrification (Figure 3.2) is the anaerobic microbial conversion of nitrate to nitrogen gases (Jacobs & Gilliam 1985). Nitrate is utilised by some microbes in the soil as an electron acceptor in the absence of oxygen during the process of respiration. These denitrifyers are heterotrophic, facultative, aerobic bacteria, that reduce nitrate to nitrite (Hill 1996). Much of the nitrite formed enters the assimilatory pathway of the nitrogen cycle, where plants and microbes in the soil convert the nitrite into protein, nucleic acids and minor nitrogenous components of the cell, and animals assimilate amino acids, building these into protein and other biological polymers (Postgate 1978). Several groups of denitrifying bacteria reduce nitrate to gaseous nitrogen forms (dinitrogen, nitric oxide, and nitrous oxide) which are lost to the atmosphere.

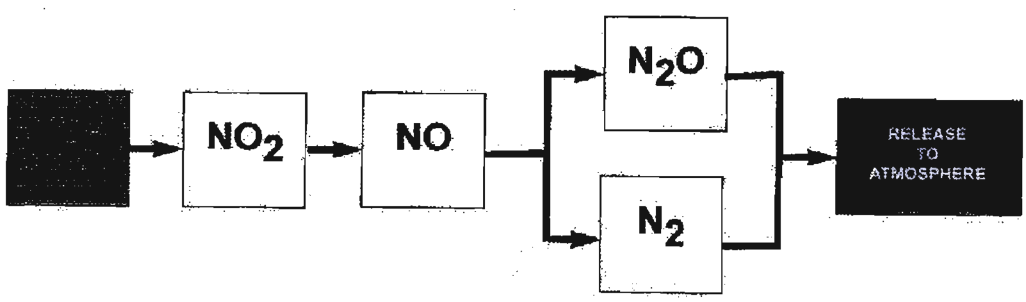


Figure 3.2: The process of denitrification, where nitrate is converted into nitrogen gas

Many researchers believe that denitrification is the primary mechanism responsible for nitrate removal in riparian land environments. Limitations in measurement techniques (Hill 1996) and spatial and temporal variations in denitrification rates in riparian ecosystems (Correll 1997), make it difficult to quantify denitrification. Researchers (Jacobs & Gilliam 1985; Cooper 1990; Ambus & Lowrance 1991; Haycock and Pinay 1993) have however, studied the organic carbon availability, pH potentials, and oxidation/reduction potentials of soils in conjunction with nitrate balances in order to surmise denitrification.

Jacobs and Gilliam (1985) inferred denitrification as the mechanism of nitrate removal from a riparian ecosystem, North Carolina, USA by the rapid declines in the NO_3^-/Cl^- ratio in groundwater. Lowrance *et al.* (1984a) also measured the chloride in shallow phreatic wells when studying nutrient cycling in the southeastern Coastal Plain, Tifton, USA. Chloride is not actively utilised by organisms, and is therefore a useful conservative marker (Osborne & Kovacic 1993). The relationship between nitrate and chloride indicates whether changes in nitrate result from dilution or evapotranspiration.

Some studies (Cooper 1990; Groffman *et al.* 1991; Pinay *et al.* 1993; Hanson *et al.* 1994) have assumed denitrification as a key mechanism through the measurement of denitrification enzyme activity. Denitrification enzyme activity assays indicate the riparian soil's potential to promote the denitrification process.

In a study of European riparian soils (Pinay & Burt 2001), denitrification rates were measured using the method of natural stable isotopic abundance. This method allowed the researchers to distinguish between nitrate removed from the system by vegetative uptake and that removed by the microbial process of denitrification. Of the two stable isotopes of nitrogen, ^{14}N and ^{15}N, the first (^{14}N) accounts for approximately 99.63% of all dinitrogen atoms in the air and is a lighter isotope than the later. The nitrogen isotopic standard of atmospheric dinitrogen remains constant, but the nitrogen isotopic standard of other materials may vary, as certain processes such as denitrification will discriminate between the two stable nitrogen isotopes, ^{14}N and ^{15}N. Denitrifying bacteria will more readily utilise the lighter isotope (^{14}N), progressively enriching the remaining nitrate pool (and the heavier isotope, ^{15}N) (Bolke & Denver 1995). Vegetative uptake will not discriminate as easily between the two isotopes (Table 3.1).

Table 3.1: The response of natural stable isotope abundance to the two processes of nitrate removal from a riparian system

PROCESS OF NITRATE REMOVAL	RESPONSE OF NITROGEN ISOTOPIC ABUNDANCE
VEGETATIVE UPTAKE	Plant uptake shows little to no discrimination of the stable nitrogen isotopes, and if this is the only means of nitrate retention in a riparian system, the isotopic composition of the residual nitrate will remain unchanged (Pinay & Burt 2001).
DENITRIFICATION	If denitrification is responsible for the removal of nitrate from a system, the isotopic composition of the remaining nitrogen will become progressively enriched, and plant isotopic composition will remain unchanged (Bolke & Denver 1995).
BOTH VEGETATIVE UPTAKE & DENITRIFICATION	If nitrate retention in the riparian system is a result of both processes, the residual nitrate will become progressively enriched, and the plant isotopic composition will reflect the nitrogen source (Pinay & Burt 2001).

At a micro-scale, three important factors influence a riparian buffer zone's ability to attenuate nitrate. These three factors directly control the process of denitrification, and will be discussed as factors influencing nitrate attenuation at an organism level. At a landscape level, various characteristics of riparian buffer zones influence the factors at an organism level. These characteristics generally relate to three important landscape factors, which will be discussed as factors influencing nitrate attenuation at a landscape level.

3.3 FACTORS INFLUENCING NITRATE ATTENUATION AT AN ORGANISM LEVEL

Henderickson (1981 as cited in Lowrance *et al.* 1997) and Jordan *et al.* (1993) measured the denitrification potential of their study sites. The general conclusion of these studies, as well as those studies by Ambus and Lowrance (1991); Lowrance *et al.* (1984b); and Jacobs and Gilliam (1985); was that denitrification occurs in most riparian land environments, and is controlled by the availability of oxygen (O_2), carbon (C) and nitrate.

3.3.1 Soil oxygen content

i) Aerobic/anaerobic conditions

The process of denitrification requires, at least for some of the time, that the soils be anaerobic, or of low oxidation/reduction potential. The processes important in maintaining a low oxidation reduction potential are composed of a series of biochemical reactions. Electrons from the organic matter released from plants, are transfered to various terminal electron acceptors (Hemond & Benoit 1988). It is the availability of these terminal electron acceptors that determines which below-ground processes (such as manganate iron reduction, denitrification, ferric iron reduction, sulphate reduction, and methanogenesis), will dominate at any one time or place in the riparian land environment (Correll 1997). The presence of molecular oxygen will inhibit any one of these below-ground processes, so these processes will only continue once the oxygen has been consumed by processes such as respiration, or sulphide and ammonium ion oxidation.

The progress of the below-ground processes is limited by a series of negative feedback mechanisms produced by the volatile end products or changing pH. The process of denitrification is an example of this. Denitrification involves hydrogen ion consumption, which results in a rise in pH. Nitrate is then converted to dinitrogen and nitrous oxide gases which are released into the atmosphere. The rates of denitrification slow down when the pH rises, whilst other processes such as nitrification increase (Postgate 1978). Thus riparian vegetation needs to have a high primary productivity to maintain low levels of oxidation/reduction potential in the soil, thereby releasing sufficient photosynthate for electrons below-ground, which will drive reactions at high rates (Correll 1997).

ii) Water inundation

Denitrification is essentially an anaerobic process, and as mentioned, requires the soil to be anaerobic at least some of the time. Fluctuating water tables are favourable for the development of vigorous coupled nitrification-denitrification activity (Hanson *et al.* 1994). Thus partial water inundation will provide the optimum environment for the process of denitrification to occur.

Permanent flooding or total water inundation however, reduces the availability of organic carbon (Correll 1997). Litter decomposition is often hindered in frequently inundated soils because the anaerobic conditions reduce the flow of carbon (energy source) to microbial populations. Denitrification enzyme activity rapidly declined from the unsaturated surface soils to the permanently saturated subsurface soils on a site studied by Lowrance (1992). The study did not detect denitrification enzyme activity below the water table, leading the researcher to conclude that permanently saturated riparian subsoils have insufficient carbon available to support active denitrification bacterial populations.

Groffman *et al.* 1992 studied the denitrification enzyme activity of a Rhode Island site, USA. There was a rapid decline in the enzyme activity from the unsaturated surface soils, to the water table and permanently saturated zone. Total water inundation was clearly implicated as a limiting factor for denitrification in this study. Similarly, Ambus and Lowrance (1991) noted a higher potential for denitrification in the unsaturated soil surfaces than in the saturated soils in a comparative study in Georgia. Yet other studies have found the denitrification process to occur in saturated conditions, even below the water table (Hill 1996). The availability of organic carbon in these particular saturated environments may have promoted denitrification. Oxygen may also be 'leaked' from the roots of vascular plants, thereby creating localized areas of oxygen availability in an otherwise saturated, anaerobic soil environment. This aerobic/anaerobic interface may enhance denitrification.

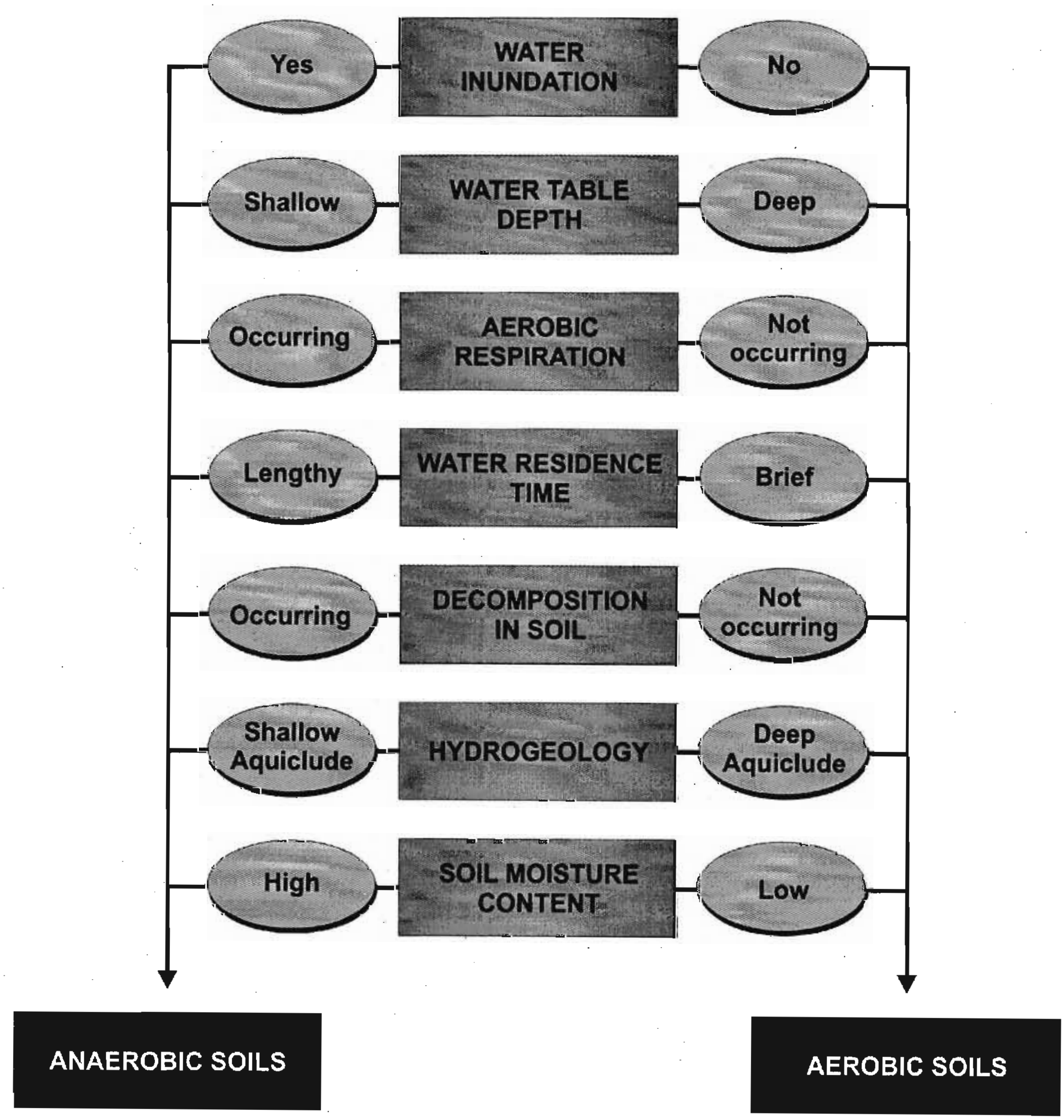
iii) Microsites

Denitrification can, however, occur in well drained soils due to the presence of anaerobic microsites, often associated with decomposing organic matter fragments which deplete available oxygen (Palone & Todd 1998; Addy *et al.* 1999). High rates of denitrification activity often occur in a patchy nature within study sites because of these localised hot spots of microbial activity (Cooper 1990; Pinay *et al.*1993; Schipper *et al.*1994). These critical 'hotspots' for denitrification exist in the subsurface ecosystems of riparian environments and may easily be missed by many studies. The nitrate removal efficiencies of many riparian sites may therefore be underestimated. Jacinthe *et al.* (1998) measured microbial nitrogen transformations in a controlled laboratory environment. They found that small patches of organic matter in the aquifer matrix will support rates of microbial activity high enough to consume available O_2, and allow anaerobic processes like denitrification to occur.

In addition to a low oxidation/reduction potential in the soil, the process of denitrification requires the availability of organic carbon, and a supply of nitrates (Haycock & Burt 1993; Hanson *et al.* 1994; Hill 1996).

iv) Factors influencing the oxygen content of the soils

Many characteristics of a riparian site will influence the oxygen content of the soils (Figure 3.3). If the soils are inundated in water, or if the water table depth is shallow, then it is most likely theat the soils are anaerobic. If the water residence time is lengthy, then the soils will remain anaerobic for sufficient time for denitrifying bacteria to turn to nitrate as an electron acceptor in the absence of oxygen. If aerobic respiration is occurring, then the soils are most likely aerobic. Aerobic respiration will utilise available oxygen, similarly the process of decomposition will utilise available oxygen, thereby leaving the soils anaerobic. A deep aquiclude will allow for deep water drainage, however a shallow aquiclude will force subsurface flows to follow shallow, lateral flowpaths, inundating surface soils in water. The moisture content of the soil most obviously indicates whether soils are anaerobic or aerobic, where high soil moisture content



reflects anaerobic soils.

Figure 3.3: Factors influencing the potential oxygen content of the soils

3.3.2 Organic carbon supply

The sustained nitrate removal by denitrifying bacteria within a riparian land environment relies on a continuous supply of carbon as an energy source (Ambus & Lowrance 1991; Haycock & Burt 1993; Hill 1996; Correll 1997). This condition for denitrification requires senescent riparian vegetation that will provide energy through decomposing leaf litter and root exudates (Lowrance *et al.* 1984b, Haycock & Pinay 1993).

Addy *et al.* (1999) studied the groundwater nitrate removal by riparian buffer zones in southern New England. Their study demonstrated that groundwater nitrate removal and denitrification gas production rates were significantly correlated with high levels of subsoil patch carbon. It has been proposed by Starr and Gillham (1993 cited in Hill 1996) that organic C within a riparian zone moves from the soil zone to shallow water table aquifers where oxidation of organic C below the water table produces reducing conditions followed by denitrification. Further to this, a study by Addy *et al.* (1999) found that microbial activity and denitrification enzyme activity in the subsoil was greater in patches of organic matter. The study indicates that total mass of patch carbon per unit of subsoil in a given riparian land environment may be a useful indicator of the denitrification potential of that subsoil. Schipper *et al.* (1994) conducted a laboratory study of microbial processes during anaerobic decomposition of plant matter, and concluded that not only the quantity of available carbon was important for denitrification, but also the quality of the available carbon. Denitrification was not dependent on total carbon added, but rather the amount of labile carbon remaining after anaerobic decomposition.

i) Factors influencing the organic carbon availability of the soils

The most influential factors in organic carbon availability are listed in Figure 3.4. Deciduous species or plants of high productivity will provide the soil with leaf litter, and high organic matter contents. The high organic matter provides organic carbon to the soil through the process of decay. High soil moisture content will allow for nutrient leaching, providing high organic carbon levels to the deeper soils.

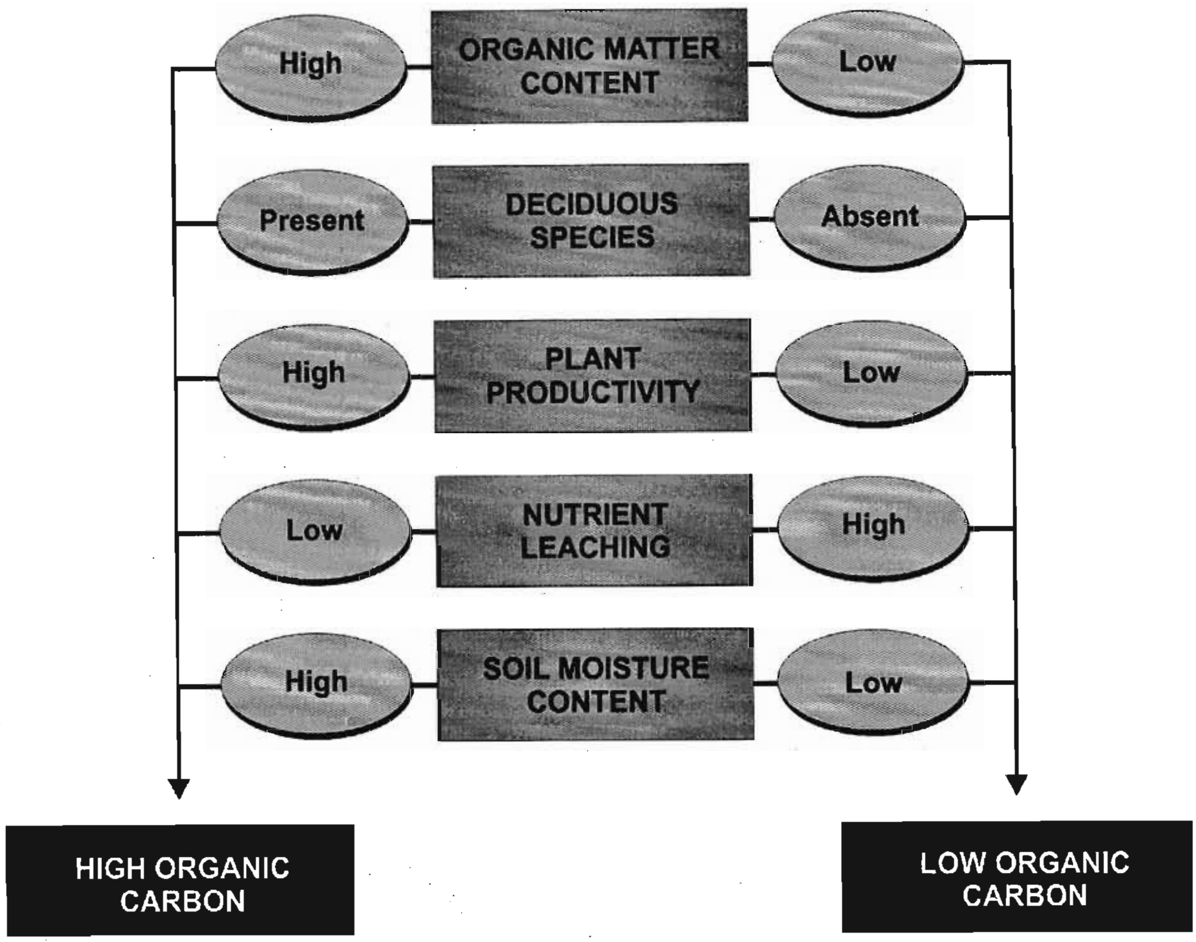


Figure 3.4: Factors influencing the potential organic carbon availability in the soil

3.3.3 Nitrate availability

Hanson *et al.* (1994) studied the denitrification potential in riparian wetlands of Rhodes Island, and determined that the interactions between riparian vegetation, hydrology and soil processes were important in the stimulation of the denitrification process in surface soils. Nitrate in groundwater was found to be utilised by the vegetation (vegetative uptake), increasing N in the plant litter and root detritus. Decomposition, mineralisation and nitrification were encouraged at higher rates by the nitrogen enrichment of litter and root detritus, which produced elevated levels of nitrate in the surface soil. Hanson *et al.* (1994) concluded that within riparian land environments, where there is a sufficient source of C and conditions are anaerobic, denitrification rates were seen to rise in response to the increased available nitrate. Haycock and Burt (1993) similarly observed that denitrifying bacteria operate best at the junction of anaerobic-aerobic zones, where both carbon and nitrate are in abundance.

Riparian areas experience rapid and frequent fluctuations of nitrate with rainfall and upland irrigation events. The Schipper *et al.* (1994) study in New Zealand on anaerobic decomposition and denitrification in organic soils furthered the understanding of microbial ecology. This study concluded that microbial populations are capable of rapidly switching metabolisms to utilise nitrate when it is available. In the absence of both electron acceptors (nitrate and oxygen) the denitrifying population were able to survive for at least 99 days, but it is undecided exactly how the bacteria were able to do this. It is possible that the denitrifying bacteria become dormant until such time as an electron acceptor becomes available, or perhaps are capable of an alternative form of anaerobic metabolism as yet unidentified (Schipper *et al.* 1994).

It was noted above that Hanson *et al.* (1994) that denitrification rates rose in response to additional nitrate, but that there had to be sufficient carbon and anaerobic soil conditions. It is suggested that nitrate will only restrict the process of denitrification under very low conditions of nitrate (< 1 mg NO_3^-N per kilogram of soil) (Altier *et al.* 1999). Denitrification is not affected by nitrate concentrations above 2-5 mg N per kilogram of soil, but is controlled by the carbon availability (Webster & Goulding as cited in Altier *et al.* 1999). For this reason, nitrate loading in the riparian buffer zone will not be considered in the mini-dissertation, as only loading rates below 1 mg NO_3^-N per kilogram of soil are influential to the process of denitrification. Loading rates below this level are not sufficiently damaging to water systems to require control.

i) Factors influencing nitrate availability to the soils

Nitrate inputs to a riparian buffer zone are largely dependent on anthropogenic sources through various land use practices, but are also possible through organism functioning (Figure 3.5). High landuse intensity and density, that is of large magnitude will provide higher levels of nitrate input to a particular catchment. Landuse intensity will indicate the intensity of the nitrate source, landuse density will indicate the number of different source areas, and the landuse magnitude will reflect the mass of nitrate per unit area. Fertilizer practices most commonly apply nitrogen, thereby increasing the nitrate availability to the riparian buffer zone. Within the buffer zone, the presence of nitrifying bacteria and nitrogen-fixing plants will convert gaseous nitrogen to ammonia, which will eventually increase the nitrate availability.

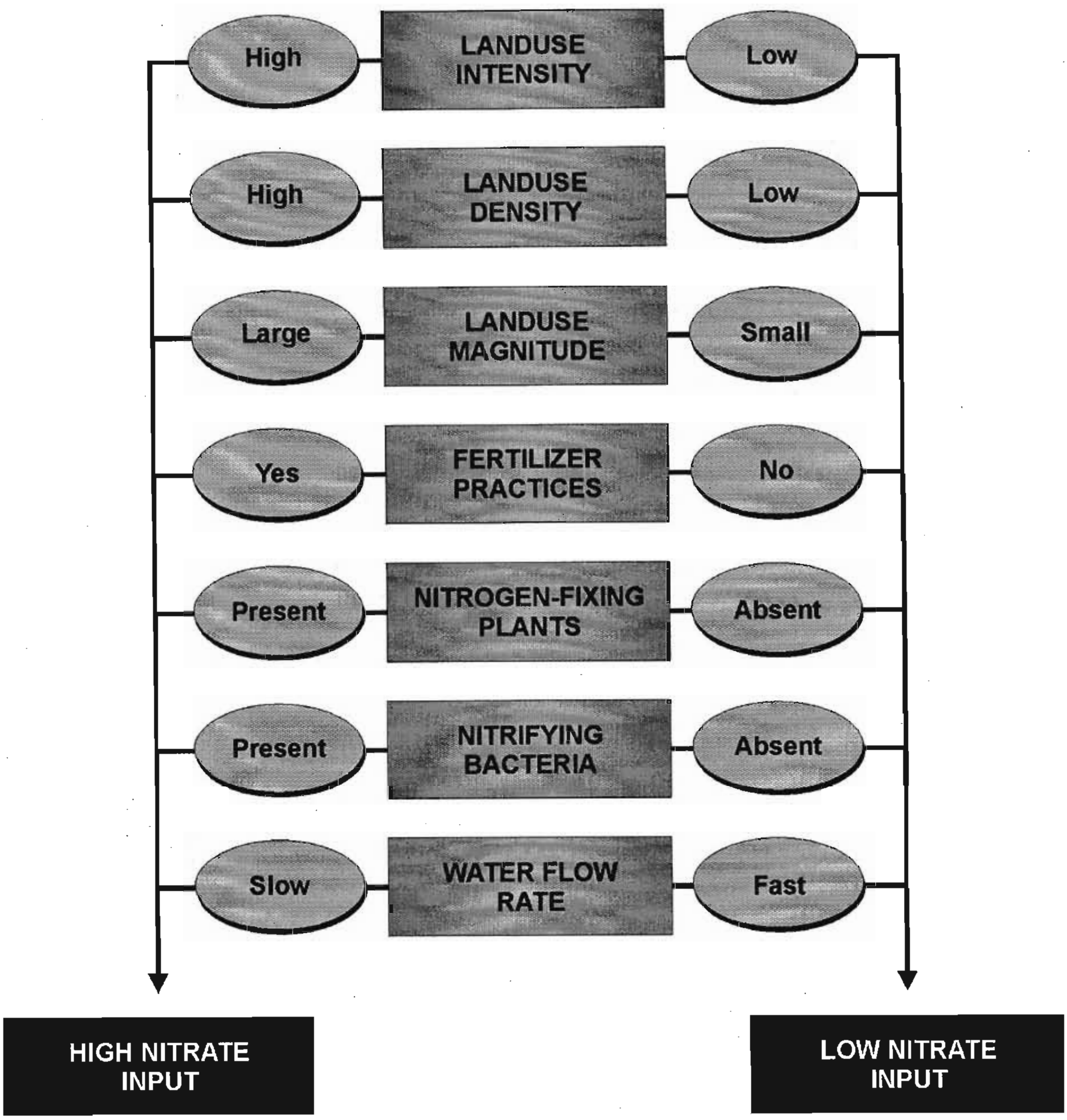


Figure 3.5: Factors influencing the potential nitrate availability in the soils

3.4 FACTORS INFLUENCING NITRATE ATTENUATION AT A LANDSCAPE LEVEL

The ability of riparian buffer zones to improve water quality is greatly affected by both internal and external characteristics of the riparian ecosystem at a landscape level (Correll 1997; Castelle *et al.* 1994). Internal or endogenous characteristics may include the width of the riparian buffer zone, the type and condition of the riparian vegetation, the nutrient inputs to the system, the geochemistry and organic matter content of the soils, the hydraulic conductivity, and the level of the water table. Whereas, the area and gradient of the watershed, the stream geology and morphology, the mineralogy and texture of the soils, the geology, and the macro and micro climates affecting the catchment, are all external or exogenous characteristics affecting the effectiveness of a riparian buffer zone (Correll 1997).

Nitrogen, or more specifically nitrate removal or immobilisation within riparian environments has generally been studied from an "input-output" perspective, where the extent of the removal or immobilisation has been based on the changes in the mass balance. Studies by Jacobs and Gilliam (1985); Haycock and Pinay (1993); O'Neill and Gordan (1994); Delgado *et al.* (1997); Nguyen *et al.* (1999); Sloan *et al.* (1999); and Lowrance *et al.* (2000b), have all measured the approximate inputs of nitrogen into the given system, and deduced the removal or immobilisation efficiencies according to the changes noted in the nitrogen outputs of the system. Little attention has been afforded to the hydrological and chemical transformation processes that produce the varying patterns of nitrate depletion (Hill 1996).

The relationship between the hydrology and chemistry of a riparian environment is however, increasingly being considered as important when understanding the role of riparian vegetation in retaining, regulating the transport of, or removing nitrate in both surface and subsurface flow (Hill 1996; Correll 1997; Lowrance *et al.* 1997). This relationship may afford an understanding of the large variations in element retention noted throughout the literature. For example, Peterjohn and Correll (1984) found that deciduous broadleaf riparian forest, 50 metres in width, retained between 47% and 90% of the incoming nitrates. That is a difference of 43%, a variation in element retention that can perhaps be explained through an understanding of the hydrology and chemistry of the study site in the Maryland catchment. The element retention rates varied according to season, and thus according to differences in precipitation, leaf litter, organic matter content and other hydrological-chemical relationships. The differences in nitrate depletion among riparian sites can perhaps be explained through these interactions of hydrology and chemistry. These interactions are to be explored in this section.

The ability of riparian zones to retain or remove nitrates has been studied in many locations around the world. Most of the riparian sites most noteworthy of nutrient retention tend to have very similar hydrogeological settings with distinct delivery pathways of water, vegetation types and soils. By comparing these studies, it is possible to isolate the common characteristics involved in nutrient attenuation in riparian buffer zones. The available studies and experiments were gathered for this mini-dissertation as part of a literature review on nitrate retention and immobilisation in riparian environments around the world. The studies were extremely varied, with different objectives, different environments; topographies and climates; different measuring techniques, and different measurement units. Often nitrate was not the main focus of the studies, and nitrate retention or immobilisation rates had to be calculated. It was necessary to standardise all the units of measurement in order to calculate the total nitrate removed from the systems. The removal efficiencies were often obscured as studies had only measured one process of nitrate removal, with little or no information on other key processes.

Table 3.2: Categories established for the data base system

NUMBER	CATEGORY	DESCRIPTION
1	IDENTITY NUMBER	Every study was numbered for easy identification
2	AUTHORS	List of authors, and date of publication
3	DURATION OF STUDY	Length of time nitrate was studied in system
4	COUNTRY / LOCATION	Specific location of study
5	RIPARIAN BUFFER TYPE	Forested; grassed; wetland or unknown
6	VEGETATION LIST	List of specific plant species found on study site
7	COVER PATTERN	Deciduous; evergreen; annual or perennial
8	COVER PERCENTAGE	Percentage of ground covered by vegetation
9	SEASON / MONTH OF STUDY	Autumn; winter; spring; summer; yearly
10	TEMPERATURE DISTRIBUTION	Mean annual temp for the duration of the study
11	RAINFALL DISTRIBUTION	Summer; winter or even rainfall distribution
12	DELIVERY PATHWAY	Main route of nitrate entry: ground or surface flow
13	SOIL DRAINAGE TYPE	Moderately well; Somewhat poorly; Poor; Very Poorly
14	SOIL CLASS	Predominantly organic; mineral or unknown
15	SLOPE	Average gradient of study site
16	GEOLOGY	Any significant geological features eg: aquiclude
17	LANDUSE ABOVE ENTRY	Type of agricultural practice above riparian area
18	NITRATE REMOVAL EFFICIENCY	Percentage of nitrate removed from site
19	DOMINANT PROCESS OF REMOVAL	Vegetative uptake; Denitrification; Both; Other
20	BUFFER WIDTH	Distance over which the nitrate was removed.

The nitrate retention rates were entered into a data base system, along with other important individual characteristics of the studies (Table 3.2). These factors included the locations, the vegetation types, the gradients, the soil types, the climate and rainfall patterns, the seasons, and the widths of the riparian environments at which nitrate retention was recorded. The purpose of this data system was to correlate all available data, and isolate similar characteristics promoting nitrate removal and retention within riparian environments. It was necessary to identify commonalities between the studies. Manipulation of the data in the form of graphs was required to better analyse these correlations. These graphs indicated the relationships between external and internal environmental factors, and the nitrate removal efficiencies of the riparian environments.

Three broad categories appeared to be very important factors in the removal efficiencies of the riparian environments around the world. The way in which the nitrate enters the riparian environment, or the delivery pathway of water tends to influence the environment's ability to remove or immobilise nitrates. Similarly, vegetation and soil factors play an important role in improving nitrate retention in riparian environments. The way in which these factors influence riparian water quality function will be discussed in greater detail below.

3.4.1 Delivery pathways

The water quality amelioration function of a riparian buffer zone is partially dependent on the volume and pathway delivery of water to the riparian ecosystem. Gilliam (1994) stated in his concluding remarks that until a better understanding of the hydrology of both surface and subsurface flows in riparian areas is obtained, it will be impossible to accurately predict the removal of pollutants by riparian buffer zones. An understanding of the hydrological characteristics within the riparian zone is instrumental in determining the effectiveness of the riparian ecosystem in retaining and removing nitrogen (Gilliam *et al.* 1997). A study of the nitrate dynamics in a small headwater stream in Scotsman Valley, New Zealand by Cooper (1990) emphasised this need to understand the hydrology of a particular catchment. The study demonstrated the important influence that pathway delivery of water and nitrate may have on the water quality amelioration functions of a riparian environment.

The delivery pathway of water to the riparian ecosystem is determined by components of the hydrological cycle, namely: precipitation, runoff and evapotransporation. Water that is not utilised in the processes of evapotransporation, or that which does not infiltrate aquifers, will enter the riparian zone from up slope as surface or subsurface flow (groundwater) either directly, or from the adjacent stream channel (Correll 1997). Some water will obviously enter the riparian zone as direct precipitation.

Figure 3.6 is a representation of many studies conducted around the world regarding the nitrate retention and removal functions of various riparian buffer systems. The graph shows the nitrate removal efficiency (as a percentage) of each study, and the width of the riparian system at which the nitrate was removed. The route of soluble nitrate entry, either as groundwater flow or surface runoff, has been indicated by the colour differentiation of each study (as depicted). There appears to be little difference in the nitrate removal abilities of the riparian systems between the two main routes of nitrate entry, other than the very low removal efficiencies noted in the groundwater study by Hanson et al. (1994). This study compared the nitrate removal efficiencies from the upland environments on either side of a small stream. The nitrate concentrations were measured six times over 31 metres and 25 metres (either side of stream). Many of the low groundwater removal efficiencies at less than 10 metres (Figure 3.6) are the findings at these intervals.

Despite the relatively similar nitrate removal abilities of riparian zones receiving ground or surface water inputs, small comparisons can be made through evidence presented in many of the findings within individual studies. The nitrate retention or removal ability of riparian systems may not be significantly affected by the route at which the nitrate enters the environments, but within each route of entry (groundwater flow or surface runoff) comparisons can be made to identify the optimum hydrogeological environment for nitrate retention or removal.

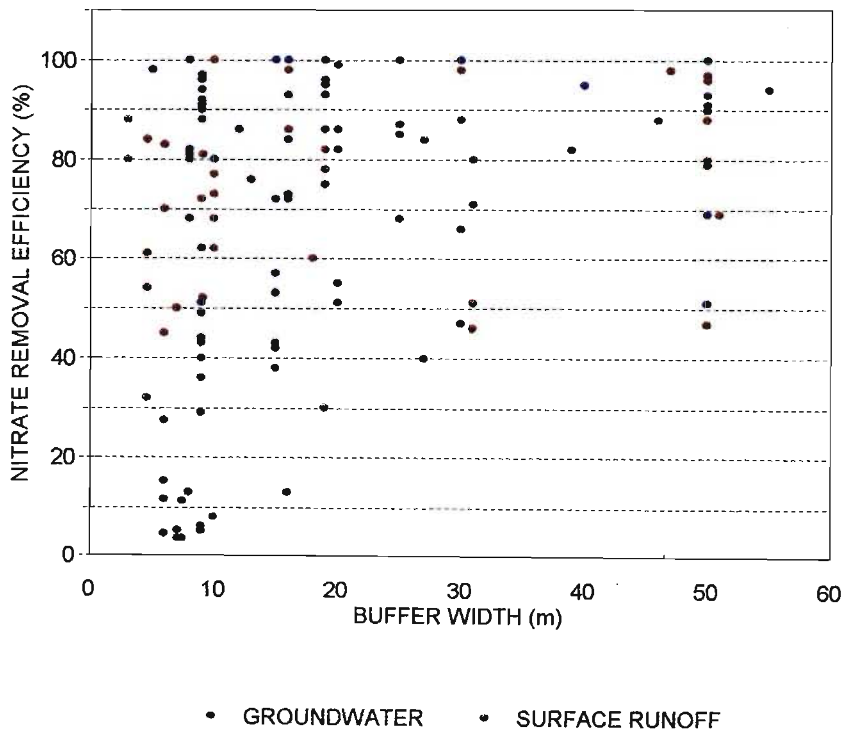


Figure 3.6: Influence of delivery pathway of water on nitrate retention of a riparian zone: the influence of route of nitrate entry on the retention abilities of riparian systems around the world

i) Surface flow delivery pathways

Surface water is most likely to enter the riparian environment in two ways. Either overland storm water runoff enters the riparian zone as shallow lateral flow, which can be both channelled or sheet flow, or the surface water may result from flood water spilling out of the stream channel (Correll 1997). Riparian vegetation can be effective in removing nitrates in both instances. Young *et al.* (1980) found that vegetated filter strips (VFS) were able to remove or retain 84% of the nitrates in surface runoff, and Dillaha *et al.* (1989) noted similar retention capacities of nitrates in surface runoff of 72% for VFS of similar width.

a) The influence of number of runoff events

Variations in the surface flow input will affect element retention capacities of riparian environments. Magette *et al.* (1989) undertook a field study in Chesapeake Bay, where simulated rainfall was used to research the nutrient and sediment removal efficiencies in surface waters by VFS. They found that VFS appeared to be less effective in nutrient removal or retention as more and more runoff events occurred. Similar findings were made by Barling and Moore (1994), and Dillaha *et al.* (1989).

Dillaha *et al.* (1989) studied the effectiveness of vegetated filter strips in nutrient retention in Virginia, USA. A rainfall simulator was used to apply water and attached nutrients to nine experimental field plots. Six rainfall events (termed 'runs') were simulated, and the nitrate concentration in the run-off exiting the plots was measured (kg/ha) at the base of each plot. Dillaha *et al.* (1989) findings are shown in Table 3.3. The first three rainfall events were summed to give a result for *runs 1-3*, as were the last three for *runs 4-6*.

For the purpose of this mini-dissertation, these results have been graphed to show the accumulated effects of rainfall events on the ability of a riparian system to retain nitrates. The graph (Figure 3.7) indicates the nitrate leaving the experimental plots for *runs 1-3* and *runs 4-6* as a percentage of the total nitrate leaving each plot over the six rainfall events. Figure 3.6 clearly indicates that a greater percentage of nitrate left the vegetated filter strips in Dillaha *et al.* (1989) experiments after repeated rainfall events. The percentage of nitrate leaving the plots increased with each event, indicating that the nitrate retention capacity of riparian environments receiving surface water flows may slowly decrease over time.

Nitrate may be deposited with the suspended sediment in the surface flow. The grass stems and stolons that provide hydraulic roughness become progressively buried (Karssies & Prosser 1999) with each runoff event. Increased runoff events over short periods of time will cause the buffer zone to reach its sediment storage capacity, and subsequent runoff events will carry the sediment and attached nutrients over the filter.

Table 3.3: Nitrate leaving nine experimental plots (Dillaha *et al.* 1989)

PLOT	RUNS	NITRATE (kg/ha)
1	1-3	0.10
1	4-6	0.26
1	**1-6**	**0.36**
2	1-3	0.37
2	4-6	1.25
2	**1-6**	**1.61**
3	1-3	0.54
3	4-6	1.11
3	**1-6**	**1.65**
4	1-3	0.36
4	4-6	1.19
4	**1-6**	**1.55**
5	1-3	0.35
5	4-6	1.50
5	**1-6**	**1.85**
6	1-3	0.75
6	4-6	1.23
6	**1-6**	**1.98**
7	1-3	0.43
7	4-6	0.79
7	**1-6**	**1.22**
8	1-3	0.08
8	4-6	0.26
8	**1-6**	**0.34**
9	1-3	0.07
9	4-6	0.27
9	**1-6**	**0.34**

If the buffer zone is unable to replenish itself with regrowth through the sediment deposition, it is possible that the nitrate retention and removal function will decrease over time, as the trapping ability of the vegetation is jeopardised by the smothering sediment. This would be in accordance with the findings of the authors above (Dillaha *et al.* 1989, Magette *et al.* 1989) regarding the decrease in nitrate removal efficiencies of riparian zones over time, and may account for some of their findings.

b) The influence of flow type

It is generally found that the riparian zones are most effective in removing nitrates from shallow, uniform surface flow derived from up slope precipitation (Jacobs & Gilliam 1985; Correll 1997). Barling and Moore (1994) state that riparian buffer strips are most effective when the flow is non-submerged (shallow), and when entry is uniform along the length of the buffer strip. This was tested in a study by Dillaha *et al.* (1989), as outlined below.

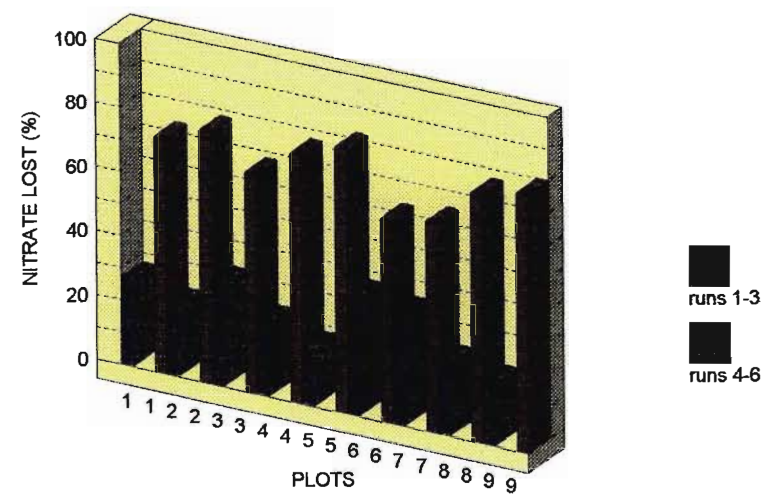


Figure 3.7: Nitrate exiting the plots for each set of runs as a percentage of the total nitrate lost from 9 plots after 6 rainfall events, as found in a study by Dillaha *et al.* (1989) . The first three simulated rainfall events are shown collectively as *runs 1-3*, and the last three are shown as *runs 4-6*

Dillaha *et al.* (1989) studied the nitrate retention of VFS of varying widths. It was found that the VFS of 4.6 and 9.1 metres retained 53% to 86%; and 70% to 90% respectively, of the sediment and attached pollutants from the shallow, uniform surface flow. The VFS of the same widths removed 83% and 93% of the sediment respectively in the concentrated flow effects (channelled flow). It was determined by the authors that the nitrate retention from the concentrated flow plots appeared to be as effective as the nitrate retention from the uniform flow plots. However, the plots were difficult to compare as the slopes of each were different, the concentrated flow plots only had a slope of 5% whereas the uniform flow plots were on slopes of 11% and 16%. These results were contradicted by Dillaha in the previous year (1988) when concentrated flow plots proved 61% to 70% less effective in nitrate retention than uniform flow plots (Barling & Moore 1994).

Daniels and Gilliam (1996) studied the effectiveness of vegetated filter strips in reducing non-point source pollution in surface runoff on two sites in Piedmont, North Carolina, USA. It was their finding that there was no effective reduction in nitrates when agricultural runoff was concentrated, however nitrate was effectively retained by the VFS when it entered the systems in sheet or rill flow (shallow lateral flow). These findings are in accordance with the findings of the authors mentioned above.

c) The influence of distance travelled by soluble nitrate

Further to the influence of flow type on nitrate reduction rates, Daniels and Gilliam (1996) found the greatest percentage of nitrate in the surface runoff was to be retained in the first few metres of the study sites (Figure 3.8).

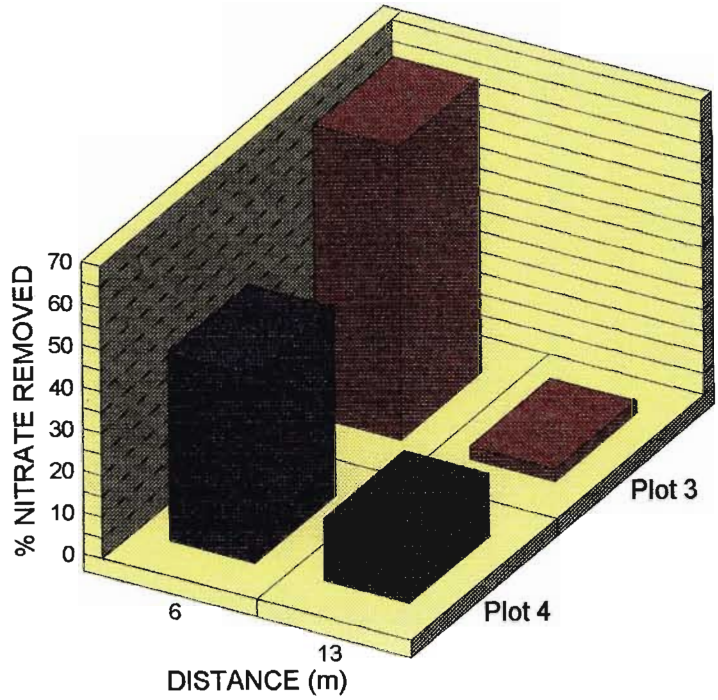


Figure 3.8: Results of study by Daniels and Gilliam (1996) indicating what percentage of the total nitrate was removed by mixed VFS at different distances from the field edge, for two experimental plots

Plot 3 in Daniels and Gilliam's study (1996) was vegetated with fescue (*Festuca arundinacea* Schreb) and groundcover, while Plot 4 had a cover of fescue, mixed hardwood and pine trees. At thirteen metres from the cultivated field edge, Plot 3 showed a 73% reduction in nitrates. 70% of this total was reduced in the first six metres. Similarly, 45% of the total 60% of nitrates removed was retained in the same distance by Plot 4. The study suggests that nitrate entering a riparian environment in surface flow is rapidly retained within the first few metres of entry. Nitrate is possibly retained in the sediment, which is usually deposited within the first few metres of the buffer zone. The reduced flow velocity caused as the surface flow reaches the vegetation promotes deposition of sediment and attached nutrients in the first few metres of entry.

ii) Subsurface flow delivery pathways

Many of the studies researching nitrate removal have observed high nitrate removal efficiencies from subsurface flows by riparian environments (Lowrance *et al.* 1984a; Haycock & Pinay 1993;

Gilliam 1994; O'Neill & Gordon 1994; Sloan 1999). The magnitude and seasonality of groundwater flows to riparian zones greatly affect the subsurface element inputs to the system (Hill 1996). The depth of the water table and the surface saturation rates of the riparian zone are directly influenced by the groundwater inputs from the higher lying areas. These hydrological factors will thus indirectly influence subsurface element retentions through the impact on the biochemical characteristics of the riparian zone. The biochemical characteristics together with the various hydraulic pathways within the riparian zone will affect element transformations (Hill 1990 as cited in Hill 1996).

Table 3.4: Studies supporting Hill's findings

AUTHOR	DATE	LOCATION
Cooper	1990	Scotsman Valley, New Zealand
Correll	1997	Edgewater, USA
Correll	1997	Edgewater, USA
Groffman	1991	Rhode Island, USA
Groffman	1992	Rhode Island,USA
Hanson *et al.*	1994	Rhode Island, USA
Haycock & Burt	1993	Cotswolds, UK
Haycock & Pinay	1993	Gloustershire, UK
Jacobs & Gilliam	1985	Beaverdam Creek, Coastal Plain, USA
Jordan	1993	Delmarva Peninsula, MD, USA
Lowrance *et al.*	1984a	Coastal Plain, USA
Lowrance *et al.*	1997	Chesapeake Bay, USA
Osborne & Kovacic	1993	Champaign County, Illinois, USA
Peterjohn & Correll	1984	Chesapeake Bay, Maryland, USA
Pinay & Decamps	1988	France
Schipper *et al.*	1993	New Zealand
Schnabel	1986	Pennsylvania
Sloan *et al.*	1999	North Carolina, USA

Further to this, differences in the residence times of subsurface flow, and the variations in hydrological pathways will also impact on element transformations within riparian zones. It is therefore necessary to explore typical geological and hydrological features contributing to these differences in subsurface flow. Hill (1996) identifies three hydrogeological settings most notably responsible for the variations in subsurface flow and delivery to riparian zones. Hydrogeological settings 1 to 3 will be expanded on below. Various studies have supported Hill's findings within each of the hydrogeological settings (Table 3.4), and will be used in the discussion of the delivery pathways of water below.

a) The influence of a shallow aquiclude

The first hydrogeological setting (Figure 3.9) promotes shallow horizontal subsurface flow. The presence of a shallow aquiclude resulting from impermeable materials underlying this setting, prevents the downward percolation of water (Correll 1997). Groundwater within this setting is therefore restricted to a local flow system, where water infiltrates upslope (recharge area), and moves as a shallow lateral flow into the adjacent riparian zone.

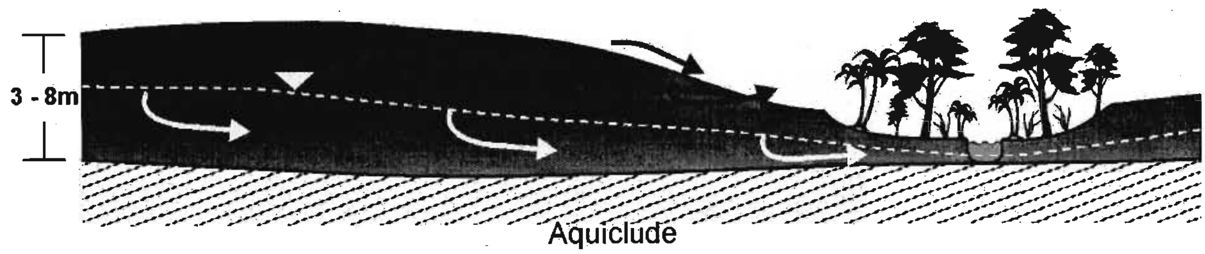


Figure 3.9: Hydrogeological setting 1 (after Hill 1996)

Many of the riparian studies on nitrate retention and removal around the world (Table 3.4) have used study sites similar to this hydrogeological setting, and generally report very high nitrate retention and removal abilities of many different types of riparian systems (Figure 3.10). Jacobs and Gilliam (1985) observed that nitrate moving in shallow subsurface water was reduced by 86%, as it travelled from the agricultural fields of North Carolina, USA, via a riparian area, into a small stream. It was further noted that a large percentage of this reduction occurred within the first 10 to 15 metres of the riparian zone. Cooper (1990) found that organic soils of a riparian area in New Zealand were capable of maintaining nitrate removal efficiencies of more than 90%, from subsurface waters flowing laterally towards the stream over an impermeable layer. Haycock and Burt (1993), and Osborne and Kovacic (1993) similarly emphasised the importance of shallow horizontal groundwater flow paths in the removal efficiencies of grassed riparian areas.

The shallow aquiclude is assumed to prevent both the percolation of nitrate down into deeper groundwater flow paths, and the movement of water upwards to dilute nitrate concentrations in lateral flowing groundwater (Gilliam *et al.* 1997). The shallow lateral groundwater flow passes in close proximity to the root systems of the riparian zone, becoming easily available for nutrient uptake by the plants. The shallow soils generally contain readily available carbon for denitrification because of the organic matter released from decaying and senescent riparian plants (Lowrance *et al.* 1984b). The fluctuating water table and surface saturation rates common to this hydrogeological setting may also promote the high nitrate removal efficiencies of riparian systems in these environments. The denitrification process is at an optimum at the junction of aerobic-anaerobic soils (Hanson *et al.* 1994), and the fluctuating soil moisture of this hydrogeological setting promotes these conditions. The variable water table may also maintain the low oxidation/reduction potential of the soils necessary for denitrification to occur (Section 3.4.3). Figure 3.10 shows the findings of many studies, where the study sites containing a shallow confined aquifer were compared to studies of other flow paths.

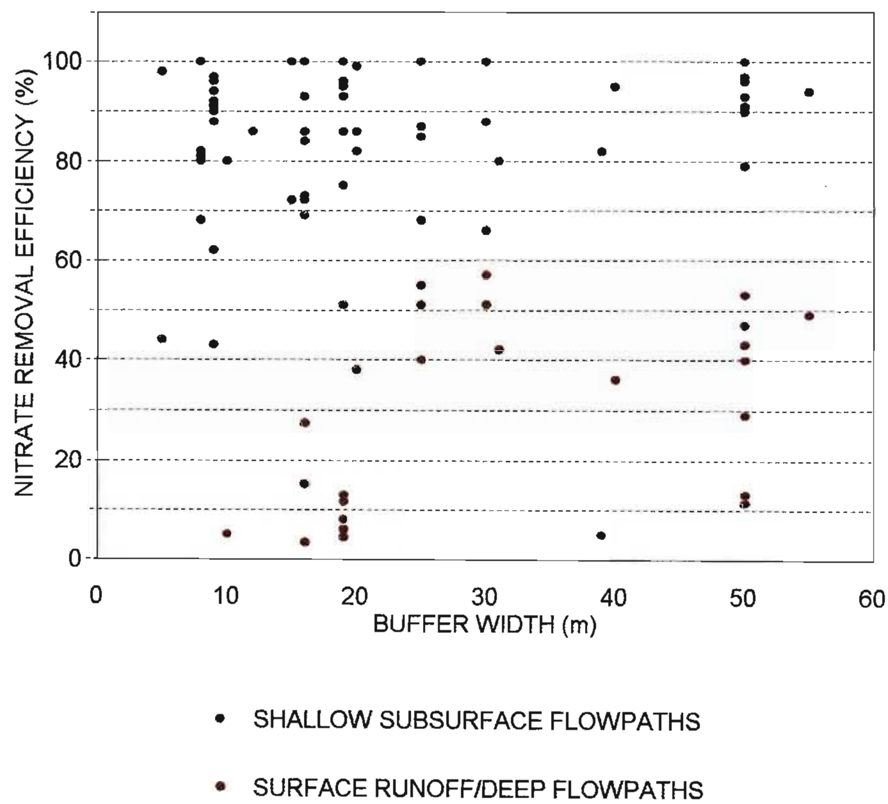


Figure 3.10: The influence of a shallow, lateral subsurface flow on the nitrate removal efficiencies of riparian zones. Created using the findings of many study sites around the world containing shallow confined aquifers

The studies where water was forced to follow shallow lateral subsurface flow paths are generally more efficient at nitrate attenuation than the other studies. This may be as a result of the close contact the nitrate has with the surface soils and vegetation in these riparian environments, where nitrate is more readily available for the mechanisms of nitrate attenuation.

The groundwater inputs to the riparian zone are limited by the shallow aquiclude, and are generally small and seasonally variable. The variability of groundwater inputs will affect the nitrate retention capacities of riparian systems. Figure 3.11 shows no clear seasonal influences in the element retention abilities of riparian zones. Five studies were used to create Figure 3.11, namely: Cooper (1990); Peterjohn & Correll (1984); Dillaha *et al.* (1989); Haycock & Pinay (1993); and Sloan *et al.* (1999). The variation in groundwater inputs throughout the seasons may account for the variability in the nitrate retention of these riparian areas.

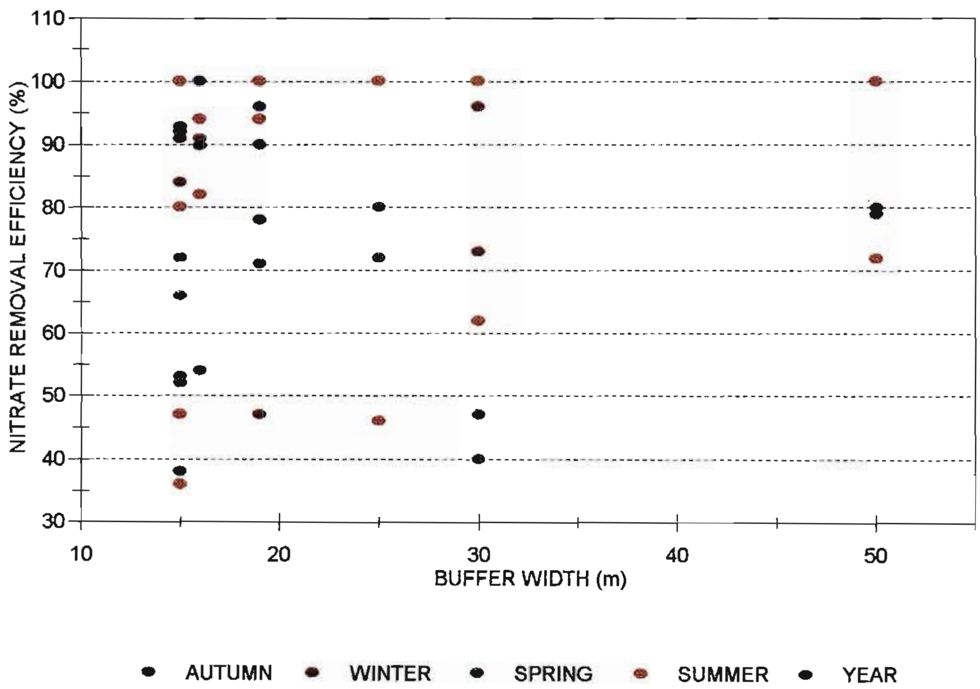


Figure 3.11: Results of five seasonal studies by Cooper (1980); Peterjohn & Correll (1984); Dillaha *et al.* (1989); Haycock & Pinay (1993); and Sloan *et al.* (1999). Graph indicates the percentage of nitrate retained by each study within each of the four seasons, and in total over the year

Peterjohn and Correll (1984) studied the nutrient dynamics within a small sub-watershed of the Rhode River drainage basin, Chesapeake Bay, USA. The site contained a perched shallow aquifer. Precipitation inputs of 27, 13, 3.5, and 44% were recorded for spring, summer, autumn, and winter respectively. Surface flow only comprised 7.1% of the annual discharge and was greatest during summer. It can be seen that the discharge to the riparian zone mainly constituted groundwater inputs, which were highly variable between the seasons. Riparian zones within this hydrogeological setting will generally have considerable fluctuations in the amount of stream flow, the depth of the water table, and the extent of surface saturation (Hill 1996). Schipper *et al.* (1993) studied denitrification and anaerobic decomposition in riparian soils in New Zealand. The study concluded that the denitrification process was possible in the winter months, but that it might be necessary to increase the proportion of flow through the sediment (as opposed to over it) for the maximum nutrient retention potential to be achieved during the winter months. Similar findings were noted by Peterjohn and Correll (1984), and Haycock and Burt (1993).

Much longer residence times of groundwater are noted within this hydrogeological setting, and therefore extensive contact with the roots of the riparian vegetation. Gilliam *et al.* (1997) noted that a longer residence time for water within a riparian environment is more effective at nitrate removal. Haycock and Pinay (1993) studied the nitrate dynamics on two sites in southern England, which contained an unconfined aquifer. The deep groundwater had a residence time of only five days, considerably less than that of the groundwater within the hydrogeological setting of a shallow confined aquifer overlying a plinthite aquiclude at a depth of 0.9 to 1.5 metres, which had a residence time of 13 to 40 days, as recorded by Lowrance *et al.* (1984a).

b) The influence of thicker surficial aquifer

The second hydrogeological setting (Figure 3.12) identified by Hill (1996) has a thicker surficial aquifer which develops in the absence of impermeable materials. Groundwater in this setting has a longer deeper flowpath to the riparian zone. Irregular land surfaces and discontinuities result in the development of a seepage face where the water table intersects the valley side. Groundwater is exposed on the surface as the water table is suddenly flattened by the upward break in the slope (Hill 1996). The surficial aquifer associated with this setting promotes deeper groundwater movement, which may discharge directly into the stream channel, bypassing riparian vegetation and soils. The limited interaction with the vegetation and soils will result in the less effective function of riparian zones within this setting.

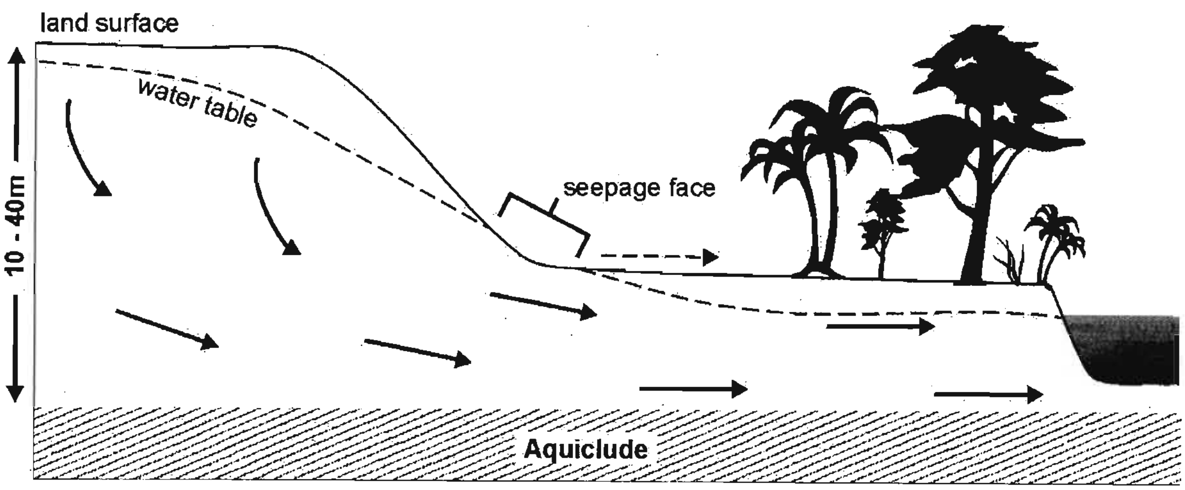


Figure 3.12: Hydrogeological setting 2 (after Hill 1996)

Haycock and Pinay (1993) compared the nitrate removal efficiencies of a poplar and a grassed site in Gloustershire, UK. Their findings were that the poplar vegetation proved more effective at nitrate retention, particularly in the winter months. The authors attributed this finding to the carbon provided to the soil by the poplar vegetation. But the vegetated sites both contained an unconfined aquifer, which may have influenced the results of their study. The deeper flowpaths created by such a hydrogeological setting may have more easily bypassed the shorter root systems of the grassed vegetation, but still would have been in greater contact with the longer root systems of the poplar vegetation.

Haycock and Muscutt (1995) determined that there was considerable spatial variability in nitrate from an agricultural landscape that moved as subsurface flow via a thick surficial aquifer. The significant amount of nitrate retained or removed within their study was spatially variable, where most retention occurred in the overlying peat sediment rather than in the deeper layers of the sand aquifer. Hill (1996) explains that large, constant groundwater inputs are received by this hydrogeological setting because of the thick extensive aquifer, which maintains a stable high water table and permanent surface saturation in the riparian zone (Hill 1996). The permanent surface saturation is not necessarily an optimum environment for the process of denitrification. Although denitrification requires the soils to be anaerobic, it also requires available carbon for energy. A permanently anaerobic environment hinders carbon availability. The permanent soil saturation and water table associated with this hydrogeological setting may therefore contribute towards the less effective nitrate removal functions noted in studies within this hydrogeological setting.

c) The influence of fluctuating surficial aquifer

The third hydrogeological setting (Figure 3.13) identified by Hill (1996) is that of a hilly upland surface. Discrete flow systems develop at different scales in the aquifer as a result of the discontinuities in the land surface. Intermediate systems are intercepted between their recharge and discharge points by local flow systems. Regional flow systems incorporate intermediate flow systems, and begin at regional topographic divides, and discharge at larger rivers (Hill 1996). The Outer Coastal Plain Flow System of Chesapeake Bay, USA is an example of this hydrogeological setting, where the plains are dominated by poorly drained soils and the ridges are dominated by well drained soils (Lowrance *et al.* 1997). Groundwater flow paths range from 1m in depth (where the aquifer is thinner) to 100m in depth, with the exception of areas adjacent to streams. In areas overlying the thicker aquifer, flow paths are up to several kilometres in length, and generally originate near the regional drainage divide (Lowrance *et al.* 1997).

It is noted that the position of the riparian zone in relation to the local and regional flow systems, and the hydrogeological setting of the riparian area within any given study, need to be understood before the water quality amelioration functions of the riparian vegetation can be quantified. These results should promote Best Management Practices to enhance the nitrate removal function efficiency, effectiveness and lifespan of riparian buffer zones.

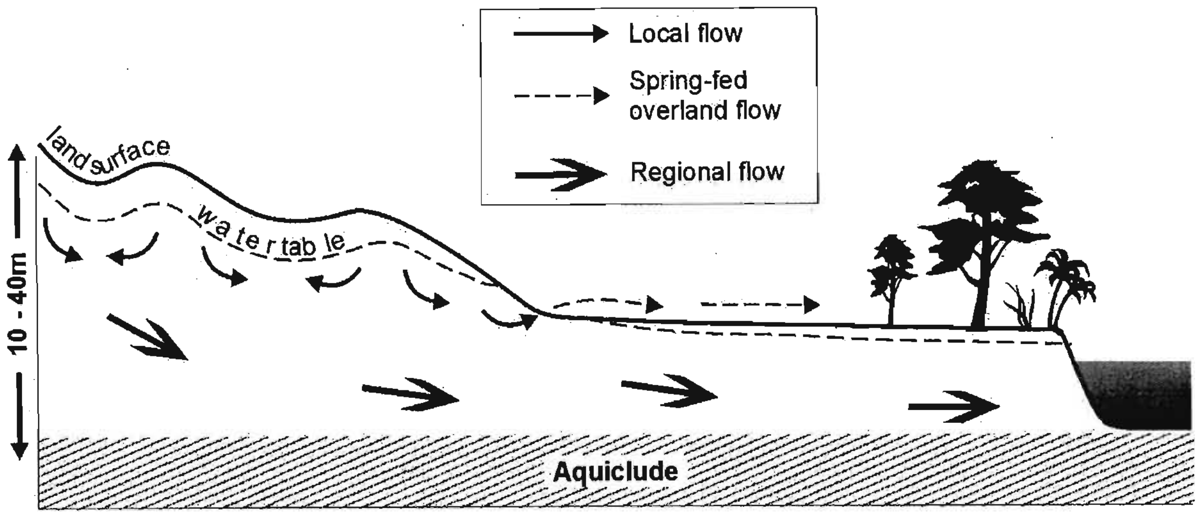


Figure 3.13: Hydrogeological setting 3 (after Hill 1996)

3.4.2 Vegetation

The choice of vegetation type for the nitrate removal function of riparian buffer zones has led to much debate. There are many studies (Groffman *et al.* 1991; Haycock & Pinay 1993; Osborne & Kovacic 1993; Schnabel *et al.* 1996; Schnabel *et al.* 1997) throughout the world relating to the topic, and most focus specifically on the nutrient retention or removal abilities of *forested* versus *grassland* buffer zones. Research to date does not allow for a definitive answer as to which vegetation type is more suited for nitrate removal or retention (Figure 3.14). Some examples of the contradictions in findings are briefly discussed below.

Osborne & Kovacic (1993) determined that both grass and forested buffer strips were effective in nitrate retention or removal, but that on an annual basis the forested filters were slightly more efficient. The nitrate entered the environments in shallow groundwater, which may account for the better results in the forested buffer zone due to the deeper nature of the root systems. Similarly, Haycock and Pinay (1993) found forested riparian buffer zones were more effective than grassed zones, especially in winter. However a study by Schnabel *et al.* (1996) contended that a grassed riparian ecotone was more effective at nitrate removal than a forested ecotone. This study indicated that the grassed riparian ecotone demonstrated greater denitrification rates than the forested site, but did expect that the forested site was carbon limited. Carbon additions to the forested riparian ecotone brought denitrification rates up to levels measured in the grassed site (Schnabel *et al.* 1996). Studies by both Groffman *et al.* (1991) and Schnabel *et al.* (1997) indicate consistently higher denitrification rates in grassed plots than rates in forested plots, but could not confidently conclude that grassed buffer zones were more effective at nitrate retention or removal than forested buffer zones.

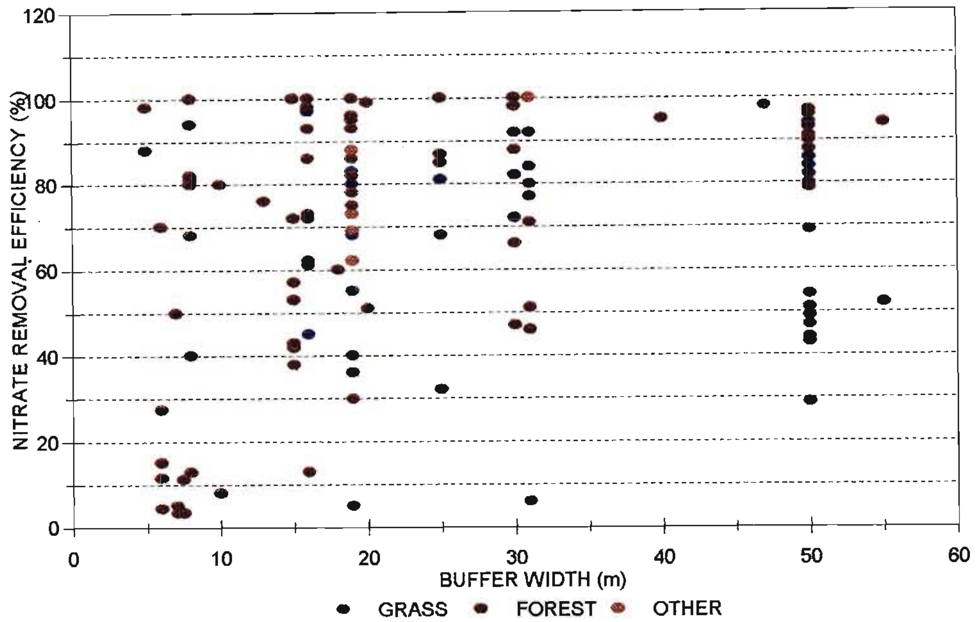


Figure 3.14: Graph indicating the effect of vegetation type on the nitrate retention or removal functions of riparian zones studied by various authors around the world

Most of the contradictions in findings can be related to differing study designs, and/or site-specific differences such as climate, hydrogeology and location. Figure 3.14 was created using findings from various studies around the world comparing forested to grassed riparian buffer zones. The percentage of nitrate removed (as determined by each study) is graphed against the width at which that percentage of removal was measured. The studies using forested riparian zones are differentiated from the studies using grassed or other riparian zones by colour. It is difficult to note the more efficient vegetation type by the graph alone. Both forested and grassed riparian zones have a wide range of removal efficiencies over a wide range of buffer widths. Both vegetation types may be both extremely efficient (>95% of the nitrates removed) or extremely inefficient (<30% of the nitrates removed) in riparian buffer zones of similar widths (eg: between five and ten metres). This finding indicates that the choice of vegetation for a riparian buffer will depend on other factors such as the soils; geology and delivery pathway of water of the environments in question.

Despite the undecided debate, some conclusions can be drawn. The main discrepancy may be related to the different routes of nutrient entry. Therefore comparisons in the nitrate removal efficiencies of the two vegetation types can be made according to whether the nitrates enter the system as surface flow or groundwater (it is acknowledged that in a given system, nitrate may enter in both flow types).

i) Vegetation and surface flow

Both grassed and forested buffer zones are effective in improving surface water quality, as they are both able to intercept or slow runoff, thereby promoting sediment deposition and infiltration (Figure 3.15). Nitrates and other solutes are made available for microbial processes and plant uptake in this way. Both vegetation types improve the storage capacity and infiltration rate of riparian soils.

Figure 3.15 indicates many studies focussing on nitrate removal from surface flow in riparian systems around the world. The nitrates removed (in percentages) are graphed against the widths (in metres) at which the percentages were measured. Neither grassed nor forested riparian zones appear significantly more efficient at nitrate retention or removal. There is also no clear indication that the width of the riparian buffer zone is of obvious importance. Both buffer types have proved extremely effective, but grassed riparian zones tend to have a few more results indicating good removal efficiencies than the forested. There are some differences noted in the nitrate removal efficiencies from surface runoff of grassed and forested vegetation types, but more clearly these differences are related to the percentage of groundcover, and the nature of the root systems in each study.

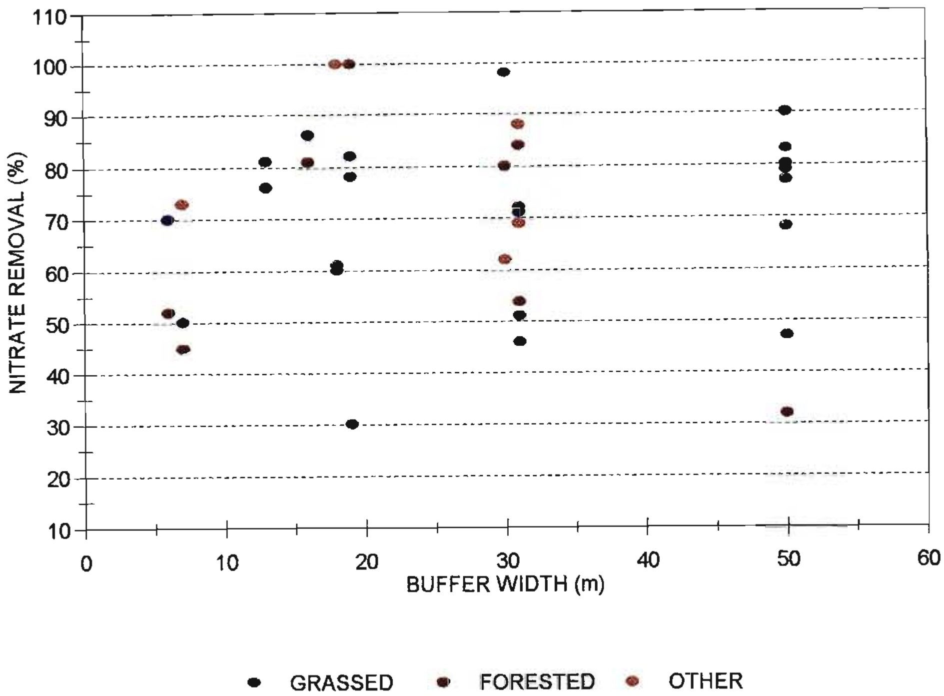


Figure 3.15: The effect of vegetation type on the nitrate removal abilities of riparian systems for nitrate entering systems attached to surface run-off. Created using findings of various studies around the world

For the purpose of sediment trapping and surface flow interception, the ground cover potential of the vegetation type is important. The greatest reductions in flow velocities are achieved by vegetation that is uniformly dense at ground level (Figure 3.16). For this reason, grassed and herbaceous riparian buffer strips tend to show greater nitrate removal or retention abilities than forested zones, as they promote a more uniform overland flow. Whereas the clumpy nature of forested sites deflects surface runoff into sparsely vegetated depressions causing concentrated flow which is more difficult to trap. If grassed zones are patchy in nature and do not provide full ground cover, surface runoff is able to bypass the grass, and the nitrate removal and retention function is rendered useless. Schmitt *et al.* (1999) studied the filter strip performance of different vegetation types and found that surface runoff was substantially slowed, and sediment deposition was substantially greater in grass and grass-shrub-tree plots than in the cultivated sorghum plots. They concluded that the numerous stems, thatch and roots of grasses were responsible for the results.

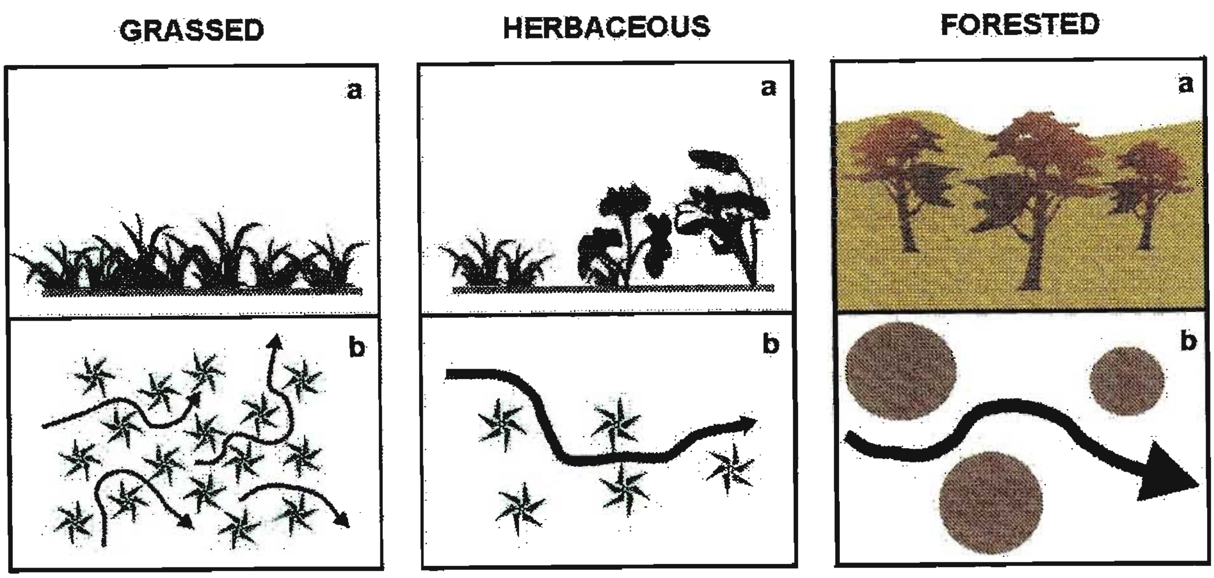


Figure 3.16: The influence of vegetation density at ground level on water flow paths (after Karrsies & Prosser 1999)

The study in Piedmont, North Carolina, by Dillaha *et al.* (1997) is one of the studies used in creating Figure 3.15. This was one of the studies showing high nitrate removal efficiencies. The groundcover in this study was between 80 and 100%, clearly influencing the trapping and filtering abilities of the grassed buffer. Another study achieving high nitrate removal percentages (as depicted in Figure 3.15) is that by Corley *et al.* (1999) in Fort Collins, Colorado which had a groundcover percentage of 99%.

The litter layer on a forest floor initially acts to trap surface runoff, but the layer may be removed by subsequent rainfall events of a more intense nature. Forested buffer zones may therefore be effective as temporary surface water trapping mechanisms. Grassed buffer zones however, may be colonised by regrowth within a few weeks, and more quickly able to resume sediment trapping abilities after they have reached their full sediment retention capacity (Karssies & Prosser 1999). Forested filter strips will be most effective at surface water retention where trees do not prevent an under storey of dense grass through shading or competition. Forests of this nature are likely in semi-arid and sub-humid environments where there is incomplete canopy cover (Karssies & Prosser 1999). Daniels and Gilliam (1996) studied a forested site with little vegetation acting as groundcover. The forested vegetation was an effective sediment and attached pollutant sink during the dry season, but ineffective during large storm events and in the wet season. The authors indicated that the little resistance to flow was the reason for these findings.

Surface flow velocity is greatly reduced by surface roughness. The height of the vegetation is therefore to be considered. Grass tends to be more efficient in this respect, as it is dense in cover and flexible in nature. Dillaha *et al.* (1989) recommend grasses between 10 cm and 15 cm in height, as they will bend over during a flow event. Corley *et al.* (1999) hypothesised that the height of riparian vegetation would affect its ability to filter and retain inorganic nitrogen. They tested two riparian communities, one dominated by grasses, and the other dominated by sedge. There was however, no consistent differences in the removal of nitrogen among the three height treatments used. There were other more important factors influencing the removal of nitrates from these riparian ecotones.

ii) Vegetation and subsurface flows

Addy *et al.* (1999) examined the groundwater nitrate removal in two similar riparian ecotones in Rhode Island, USA. The sites differed in that one was forested (woody) vegetation and the other was mowed (herbaceous). They observed substantial groundwater nitrate removal and denitrification at both locations, with no significant difference between the two vegetation types. In respect to these findings, Addy *et al.* (1999) caution against ascribing specific groundwater nitrate removal rates to differing aboveground vegetation without recognising site specific differences (water table dynamics, land use legacy, and adjacent vegetation).

Figure 3.17 indicates many studies focussing on nitrate removal from groundwater flow in riparian systems around the world. The nitrates removed (in percentages) are graphed against the widths (in metres) at which the percentages were measured. Neither grassed nor forested riparian zones appear significantly more efficient at nitrate retention or removal. There is also no clear indication that the width of the riparian buffer zone is of obvious importance. It is evident that site specific differences other than vegetation type also influence the nitrate retention capacities of riparian zones.

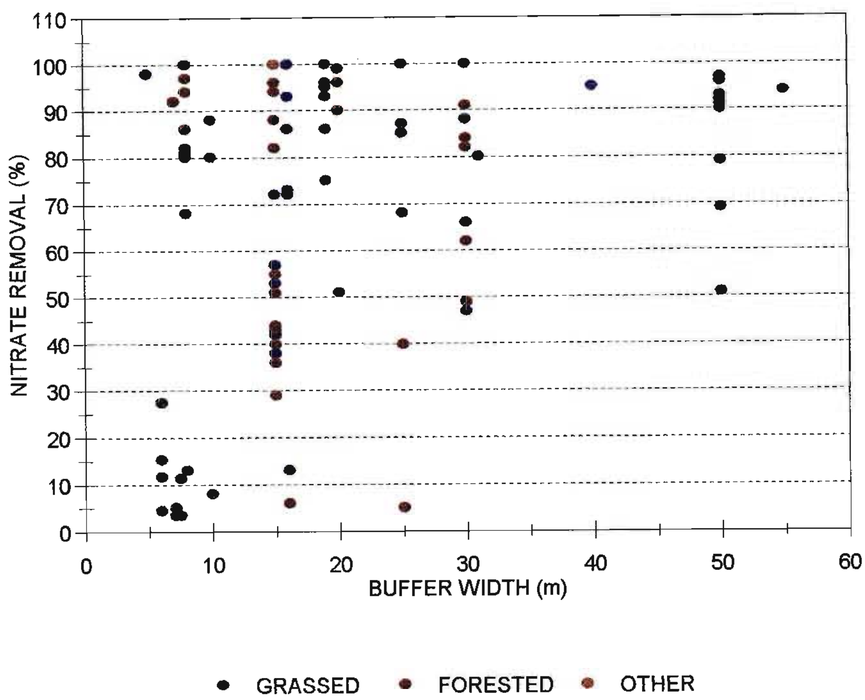


Figure 3.17: The effect of vegetation type on the nitrate removal abilities of riparian systems for nitrate entering systems attached to groundwater flows. Created using findings of various studies around the world (subset of studies used in Figure 3.14)

Yet some studies have identified certain factors responsible for nitrate retention differences in the two vegetation types. Haycock and Pinay (1993) indicated that a poplar-vegetated site in southern England was more efficient than a grassed site in groundwater nitrate retention, especially during the winter months. Both sites were extremely successful at nitrate retention, where the grassed site retained 84% of the nitrates entering the system, and the poplar site retained 99% of the nitrates entering the system. But the poplar site was significantly more effective during the winter months, when aboveground biomass is inactive. Woody vegetation associated with forested buffer zones is able to supply organic matter to the soil, and is more effective at supplying organic matter to the deeper subsoils, where it maintains a low oxidation/reduction potential and is made available for the process of denitrification (Correll 1997). The authors expect that the aboveground biomass still contributed sufficient carbon to the soil to allow for microbial processes such as denitrification to occur. Shrub and forested riparian buffer zones tend to have more varied root systems and depths than grassed buffer zones. Shallow groundwater flow is more easily intercepted by these root systems, and either drawn to the microbially active layer by the capillary action of the plants, or utilised by the plants themselves. The forested site in this study was therefore more efficient in nitrate retention and removal. The carbon availability was also noted by Haycock and Burt (1993).

Groffman *et al.* (1991) measured the rates of denitrification in two grass and two FBS in a study in Rhode Island. The authors expected the forested vegetation to have more potential for the process of denitrification than the GFS. This was anticipated as forested sites tend to have higher soil moisture contents and organic matter levels than grassed sites, both factors influencing denitrification. Yet the authors found the GFS to have consistently higher denitrification activity in response to added nitrates than the soils from the FBS. It was noted however that the GFS had received applications of lime and fertilizer, which may have made the soils more favourable for denitrification. Further to this the study showed that tall fescue grass was more effective at nitrate retention than red canary grass. These results may be because of the carbon inputs from the roots to the soil, which differs between the grasses.

In a study of 13 European sites (Pinay & Burt 2001), the herbaceous sites showed significantly higher rates of nitrogen re-translocation and decomposition when compared to the forested sites. This was attributed to the differences in litter composition such as ligneous leaf veins. Overall, there were no clear differences in nitrate retention capabilities between the herbaceous and the forested sites, but the higher nutrient cycling of the herbaceous sites may improve the potential for denitrification and nitrate residence times. Yet there were no significant differences in the denitrification rates between the herbaceous and the forested sites in this study (Pinay & Burt 2001), suggesting that vegetation type was less important than other factors such as the delivery pathway of water to the sites.

3.4.3 Soils

Riparian areas usually have very varied soils where local weathered soils; deposited soils; and organic debris are found to be in combination within these environments. Certain features of the soil affect the way and rate at which the water flows over and through a riparian area, the extent to which the ground water remains in contact with the plant roots and soil particles, and the degree to which the soils become anaerobic. These features of the soil which influence the water quality amelioration function of a riparian area include soil texture, structure, permeability, porosity, chemistry, organic matter content, and the depth to the water table (Klapproth 1999, Palone & Todd 1997).

i) Soil texture

Parent material directly affects soil texture, which is the proportions of clay, silt and sand below 2mm in diametre contained in mineral soil. Organic soil textures are classed as muck, peat, mucky peat, and peaty muck (Palone & Todd 1997). Soil texture influences the chemical properties of the soil, soil moisture content, and root development. The soil texture of a riparian buffer zone will therefore influence the type and condition of vegetation that will grow. The nitrate removal or retention function of the riparian buffer zone will be affected by the vegetation, soil moisture content, and chemical properties of the soil. Coarse textured soils are more aerated than finer clays (Palone & Todd 1997). Soil oxygen content influences the process of denitrification.

Hubbard and Lowrance (1996) state that soil texture affects water movement as it generally moves faster through coarser textured materials than through finer textured materials. In their study on vertical and lateral transport of water and attached pollutants in the Coastal Plain of the USA, they concluded that consideration should be made of underlying soil and geological materials and anticipated rate of water movement before designing management systems which utilise riparian systems for nitrate and other nutrient retention and removal.

The Center for Environmental Policy at the University of South Carolina, USA (Final Report, RFB 2000) state that soils of high clay content are ineffective for water quality functions of riparian buffers as their permeability is too low. However mixed clay soils are more effective as they retain water for longer periods of time, have higher organic matter contents, and promote growth and maintenance of microbial communities, thus allowing for greater pollutant removal. Mixed clay soils act as anions (negatively charged particles), and have a high affinity for binding positively charged particles, especially metals. Pollutant removal via this mechanism may be significant if the soils are not compact and run-off is slow (Final Report, RFB 2000).

ii) Water table

The water table is the upper most level of saturation in the soil, which demarcates the top of the groundwater. It is the location at which the soil water is at atmospheric pressure. Soil water below the water table is subject to hydrostatic pressures, while the soil water above the water table is under tension from capillary suction (Gold & Kellogg 1997). As riparian environments extend laterally from the upland boundary to the water body, they are usually of more gentle gradient and therefore on sites which promote shallow water tables. The depth of the water table has many implications for the internal processes of a riparian environment.

Shallow water tables can effect the extent of water extraction by vegetation. In the upland areas, soil moisture declines in response to evapotranspiration, as it is not replenished by groundwater, which makes its way down the slope. But in a riparian buffer zone with a shallow water table, soil moisture removed by vegetation is replaced by the tension-free groundwater. Evapo-transpiration generates an upward flow of water to the root zone, termed 'upwelling'.The potential for upwelling results from the gradient between the drying soil in the root zone and the tension-free groundwater (Gold & Kellogg 1997). The rate of upward water movement is determined by this matrix gradient, the depth of the water table, and the unsaturated hydraulic conductivity (Altier *et al.* 1999). Nitrates and other soluble pollutants in groundwater arrive in the biologically active zone of the riparian soil through this process of upwelling, where they are made available for plant uptake and microbial activities. The presence of a shallow water table is therefore influential in the removal of nitrates from a riparian buffer zone.

The study of 13 European sites in the Nicolas Project (Pinay & Burt 2001) showed the level of the water table to be one of the most influential factors in denitrification rates. In sites where there was a high groundwater table of above -30 cm, denitrification rates were at their highest. The high denitrification rates in the waterlogged surface soils were not related to soil temperature or soil grain size. It was evident that the level of the water table controlled the rates of denitrification. But in dryer sites, denitrification rates were dependent on soil grain size. It was found that rates of denitrification were only high when silt and clay contents were high. It is suggested that sites of fine grained soils maintain anaerobic conditions for longer periods of time after rainfall events (Pinay & Burt 2001). The level of the water table conditions the mineralisation process and its end products, and the study by Pinay and Burt (2001) indicates this prevalent role the water table plays in denitrification, above other factors such as climate and vegetation type.

iii) Soil moisture and oxygen content

Soil moisture can be expressed in various formats. Gold & Kellogg (1997) express soil moisture on a volumetric basis and as a water-filled pore space percentage (Table 3.5).

Table 3.5: Soil moisture as expressed by Gold & Kellogg (1997)

FORMAT	FORMULA	DEFINITION
Volumetric basis	$\theta_v = V_w / V_t$	θ_v = volumetric water content (cm^3/cm^3) V_w = volume of water (cm^3) V_t = total volume of soil (cm^3)
% Water-filled pore space	**WFP = θ_v/ϕ * 100**	WFP = % water-filled pore space ϕ = total porosity (cm^3/cm^3)

The ability of a riparian environment to control the quality of surface and ground water entering a stream system is partially as a result of the unique properties found within the upper 1-2 metres of a riparian soil. This layer is referred to the biologically active zone, and extends to where the soils show evidence of periods of saturation (Gold & Kellogg 1997). The temporal and spatial patterns of soil moisture within the biologically active zone affects the redox potential of the soil, the infiltration rate and root uptake potential, evapotranspiration, microbial activities, and the extent and the timing of water table fluctuations. As nitrates are removed or retained as solutes in the soil water, the soil moisture content will greatly affect the nitrate removal function of a riparian buffer zone. Linn and Doran (1984 as cited in Gold & Kellogg 1997) studied the relationship between microbial activities and the water-filled pore space (WFP) in soils. It was found that many microbial processes, both aerobic and anaerobic, were influenced by the WFP. The relative denitrification rate within a soil increases five-fold when WFP increases from 80% to 100%. The microbial activity was more sensitive to WFP in the range of 60% to 100% (Gold & Kellogg 1997).

Correll (1997) contends that the denitrification processes responsible for nitrate removal require that the soil be anaerobic for at least part of the year (biological denitrification is carried out by anaerobic microbes, and requires that soils be wet enough for oxygen depletion to occur). Similarly, Schnabel *et al.* (1996) studied rates of denitrification in riparian ecotones in Pennsylvania, and noted that denitrification appeared to be controlled by soil moisture content. Soil moisture content was greatest near the stream, and caused greater resistance to oxygen diffusion. The resulting larger volume of anaerobic microsites near the stream corresponded with increased denitrification rates. A comparative study on nitrate removal by Hanson *et al.* (1994) indicated that a nitrate-enriched site with high levels of soil moisture relative to the control site, was also more effective at removing nitrates. It was concluded that the increased soil moisture promoted the process of denitrification.

iv) Soil drainage

The frequency and duration of periods of saturation or partial saturation during soil formation is referred to as natural drainage (Palone & Todd 1997). Of the seven classes of soil drainage used to describe or define soils, four (as described by Palone & Todd 1997) will be discussed here:

- *very poorly drained(VPD):* the water table remains at or on the surface of the soil for much of the time, and water is therefore removed very slowly. VPD soils are common in depressed sites, where water accumulates, and are frequently ponded.

- *poorly drained (PD)*: soils remain wet for a large part of the time as water does not infiltrate easily, or because of seepage. The water table remains at or on the surface for a considerable part of the year.
- *somewhat poorly drained (SPD):* a slowly permeable layer within the soil profile, a high water table, and additions through seepage or rainfall are common in soils of this drainage class. Water is slowly removed from the soil, and the soil is kept significantly wet, but not all the time.
- *moderately well drained (MWD):* the profile is wet for a small, but significant part of the year, as water is removed from the soil somewhat slowly. Soils of this drainage class commonly have a slowly permeable layer within or immediately beneath the solum, a relatively high water table, or additions of water through seepage.

Moderately well drained soils are able to intercept large amounts of surface flow promoting deposition of sediment because they have the greatest permeability. This is important for nutrient trapping as sediment-attached pollutants are deposited too, and made available for long periods of time. Yet in terms of nitrate retention or removal, fine-textured soils that are somewhat poorly to poorly drained, create more favourable conditions for denitrification (Palone & Todd 1997).

Rates of denitrification in two riparian forest sites in Rhode Island were studied by Hanson *et al.* (1994). The sites were similar in vegetation, slope and soils, but differed in upland landuse and nitrate inputs. Each site contained an upland to wetland transition zone, with moderately well drained (MWD) and somewhat poorly drained (SPD) soils of the upland becoming poorly drained (PD) and very poorly drained (VPD) in the wetland. The rates of nitrate removal on both sites were higher at the wetland end than at the upland edge, indicating that the soil drainage may have been influential in nitrate removal efficiency. The higher soil moisture and organic matter contents of the PD and VPD soils favoured the process of denitrification, but the MWD and SPD soils were also noted as significantly efficient in nitrate removal. These soils were transition soils between the wetland and upland, with fluctuating water tables favouring nitrification-denitrification coupling activity (Reddy & Patrick 1984 as cited in Hanson *et al.* 1994).

Riparian buffer zones provide decaying organic matter to the soil for oxygen depletion to occur (Cooper *et al.* 1995). Organic matter also supplies carbon as an energy source to fuel the denitrification process. Even drier soils are therefore able to reduce nitrates to some degree as 'pockets' of organic matter and microbial activity develop within them (Palone & Todd 1997).

Shallow surface soils allow maximum nutrient uptake by plants as the roots are in close proximity to the depth of the water table. Soil drainage should be considered in riparian zone protection and management for nitrate removal.

v) Organic versus mineral soils

Organic soils are classified as having at least 20% organic matter (by weight) in soils of low clay content, or at least 30% organic matter (by weight) in soils of high clay content (>60%) (Palone & Todd 1997). Mineral soils consist predominantly of, and have properties determined predominantly by mineral matter. These soils usually contain less than 20% organic matter (Palone & Todd 1997). When considering the nitrate removal efficiencies of soils, organic soils are found to be more suited for denitrification of incoming nitrates than mineral soils. This is perhaps because organic soils are anaerobic, have high concentrations of microbially-available carbon, and possess a high enzymatic capability to denitrify (Cooper 1990). In a study in New Zealand by Cooper (1990), organic soils comprised 12% of the soils bordering a stream, but were responsible for between 56 and 100% of the nitrates removed from the riparian-stream system (Figure 3.18). It was noted that a disproportional amount of groundwater flowed through the organic soils enhancing their role in nitrate trapping and removal. Organic soils are still, however capable of infiltrating large amounts of surface runoff and have a high affinity for nitrogen and other contaminants. Organic soils in riparian areas therefore promote nitrate removal. Figure 3.18 shows the percentage of nitrate retained by the riparian zone per each month of study. The two soils types (organic or mineral) are differentiated by colour, as depicted. It is evident from the graph, that a greater percentage of nitrates was removed from flow systems entering the organic soils, than from the mineral soils in the study. The organic soils promoted nitrate retention.

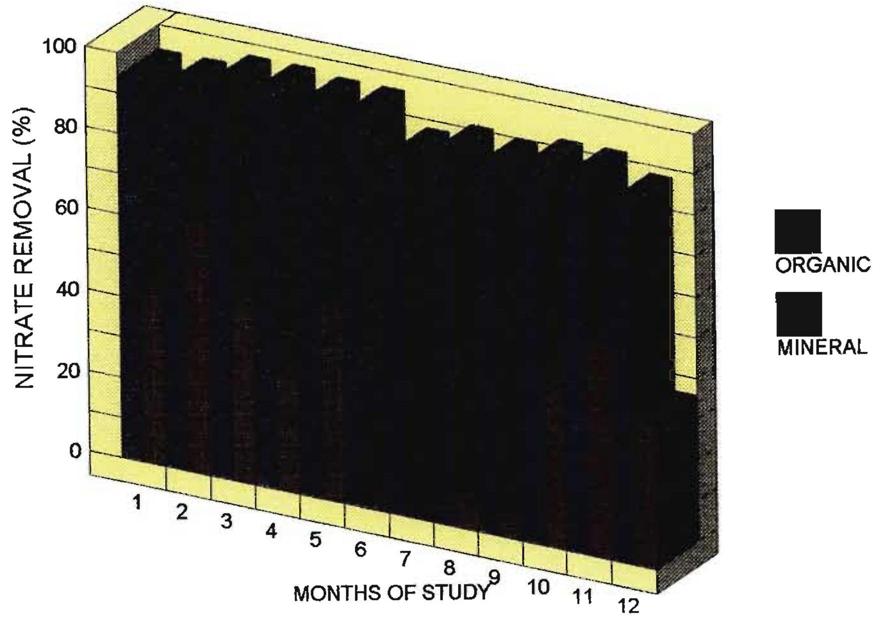


Figure 3.18: The effect of soil type (organic or mineral) on the nitrate removal efficiency of a riparian zone in New Zealand, as found by Cooper (1990). The study was conducted over a period of 12 months

vi) Microbial activity

Soil microorganisms are responsible for much of the water quality amelioration functions of soils, as they remain constantly active. The microorganisms utilise or convert nutrients into forms which are less biologically available and more readily stored in the soil (Klapproth 1999). They utilise organic chemicals as an energy source, and require ions for the process of transforming these chemicals into less toxic compounds.

vii) Slope

Sediment removal, transport and deposition are greatly influenced by the degree of slope of a particular site. Generally, gentle slopes promote infiltration and water retention and allow nitrate to be available for microbial processes for longer periods of time, whereas steep slopes enhance runoff velocity and reduce probability of deposition. Buffer zones may therefore need to be increased in width to improve nitrate removal potential if the site is steep in nature.

In the Nicolas Project (Pinay & Burt 2001), which studied 13 sites throughout Europe, it was noted that slope and river water contributed towards the saturated conditions of the riparian zone. The authors identified that the potential for denitrification increases towards the soil surface, and therefore suggest that the water table will influence the degree to which the nitrate reduction on the site is optimised. Hence the considerable influence the slope will have on the behaviour of the water table will greatly affect the site potential for nitrate removal.

The study found that in sites where the river was bordered by sloping land, the behaviour of the water table was greatly influenced by the upland conditions, but the river's influence on the soil conditions was limited. The water table on sloping sites remains almost parallel to the groundsurface throughout the year, regardless of its depth (Pinay & Burt 2001). The authors anticipate that on such sites, especially where soils are deep, permeable and confined by an aquiclude, there will be significant subsurface flow within the soil. The process of denitrification will be more effective as more water will slowly pass through the soil parallel to the ground surface. Very steep slopes may promote high rates of runoff and subsurface flow, and it is therefore estimated that slopes of 5 to 10 degrees will be most effective at nitrate removal. The Centre for Environmental Policy at the University of South Carolina (Final Report, RFB 2000) report that many findings indicate that a slope of greater than 15% will not allow for sufficient run-off retention time, and will therefore be ineffective at sediment trapping and pollutant removal. Even well vegetated slopes of greater than 15% may promote rill flow, channelization and potential for erosion. It is recommended that a slope of less than 15% will slow run-off, promote sheet flow and infiltration, and allow for pollutant removal by vegetation and denitrification.

When a riparian area is bordered by flat land or little to no slope, such as a floodplain, the river water will have more influence on the soil conditions. The water table behaviour will be more influenced by the river conditions. The water table will fluctuate seasonally, according to the upland inputs and the river level (Pinay & Burt 2001).

4. TOWARDS MANAGEMENT RECOMMENDATIONS FOR THE ATTENUATION OF NITRATE

4.1 INTRODUCTION

When a specific management objective of water conservation is nitrate attenuation from agricultural surface and subsurface flows, then riparian buffer zones may offer an economical and sustainable conservation practice. The two key processes (vegetative uptake and denitrification) responsible for nitrate attenuation within buffer zones should be promoted. The previous chapter explored three influential factors in nitrate attenuation at a landscape level in riparian zones world-wide, namely: the delivery pathway of water; the vegetation; and the soils. The rehabilitation, design and maintenance of riparian buffer zones for nitrate attenuation will require that these influential factors be enhanced or sustained.

Many of the characteristics of these three factors within riparian buffer zones which best promote nitrate attenuation have been established in Chapter 3. This chapter proposes that these characteristics can be used to both assess a riparian landscape for its potential to attenuate nitrate, and to size a riparian buffer zone specifically to meet this function.

4.2 GUIDELINES FOR ASSESSMENT AND WIDTH DETERMINATION

Riparian rehabilitation or creation can be an extremely daunting task, especially when large reaches of river have been degraded by poor land management or neglect. It is often difficult to determine not only where to place riparian buffer zones, but also how to start rehabilitating; revegetating or improving the chosen sites. Certain models have been developed at a catchment level, that have integrated the results of individual studies into tools useful for planning and management of riparian buffer zones. For example the Riparian Ecosystem Management Model (REMM) is a computer simulation model of a riparian forest buffer systems, developed to allow the user to investigate different scenarios. The model is very useful, but requires large inputs, with knowledge of upstream hydrology, and therefore technically difficult to use at a small farm scale. Guidelines based on the results of individual nitrate attenuation studies within a South African context are lacking.

A set of guidelines have been developed as based on the findings of this mini-dissertation, and are designed as a proposed means to assist farmers and similar landowners in riparian buffer zone creation for the retention or removal of nitrate. The guidelines are set out as three steps (Table 4.1). The first two steps of the guidelines can be used to set prioritised management zones, and then help assess the site potential for nitrate attenuation. If a management zone is of high priority, and has a poor potential for nitrate attenuation, then establishing riparian buffer zones may be a good conservation practice for nitrate attenuation. Although there is a large literature base on how to establish riparian buffer zones, there is little consensus on how to size a riparian buffer zone specifically for nitrate attenuation. The third step to the guidelines proposed in this chapter can be used to estimate a more accurate buffer width for a specific management zone. Finally, the chapter explores a case study, to show the practical application of the guidelines.

Table 4.1: Proposed guidelines for assessing and sizing riparian buffer zones specifically for nitrate attenuation

GUIDELINES	
STEP 1: **To establish prioritised management zones**	Step 1a: Delineate the catchments Step 1b: Identify the nitrate loading potential for each management zone Step 1c: Identify delivery pathways in each management zone Step 1d: Prioritise management zones
STEP 2: **To assess nitrate attenuation potential in each management zone**	Step 2a: Establish denitrification potential Step 2b: Establish vegetative uptake potential Step 2c: Score nitrate attenuation potential
STEP 3: **To determine adequate buffer width**	Step 3a: Score organic matter content rating Step 3b: Score the hydrologic soil group rating Step 3c: Score the slope rating Step 3d: Score the subsurface flow rating Step 3e: Achieving a width estimate

4.3 PRIORITISING AND ASSESSING MANAGEMENT ZONES

<u>**STEP 1. TO ESTABLISH PRIORITISED RIPARIAN MANAGEMENT ZONES**</u>

With nitrate attenuation as a management objective, the first logical step would be to divide the given agricultural area into smaller, more-manageable units, and to prioritise sections of river for rehabilitation or buffer zone creation. The nitrate loading and delivery pathways of water will give a better indication of management zone priority.

Step 1a) To delineate the catchments

The first part of ***Step 1*** requires that the micro-catchments be defined. With topographical maps and orthophotographs of the agricultural area, the drainage lines and watersheds can be marked using either manual or GIS techniques. As nitrate is a solute, it will reach the river system either in surface or subsurface flow (Prosser *et al.* 1999c). The micro-catchments will give a better indication of the where the water is coming from, as well as the direction of the flow and the way in which it enters the river system. Because water in each micro-catchment has a common watershed and drainage area, the micro-catchments can each become individual management zones.

Step 1b)   To identify the nitrate loading within each management zone

Nitrate in agricultural run-off is most often derived from non-point sources. The non-point source nitrate characteristics will give an indication of the potential nitrate loading for each management zone. It is assumed that the agricultural landuse is immediately adjacent to the river system. The nitrate loading refers to the mass of nitrate per unit area, which is usually measured over a period of time. The solute nitrate will move through the drainage system, and the loading can ultimately be measured at a drainage point in kilograms per litre. Although the actual loadings can be calculated given sufficient information, the approach taken for the purpose of this research is to obtain a relative measure of loading.

For example, the extent of the landuse within each management zone can be used to gain a relative measure of the potential loading rates, where source density, intensity and magnitude can be rated (Table 4.2). Source density might be the number of different source areas in the management zone, source intensity might be the frequency of fertilizer applications, and source magnitude might be the mass of fertilization per unit area.

Table 4.2: Example of how to establish nitrate loading potential

	HIGH	MEDIUM	LOW
DENSITY	3	2	1
INTENSITY	3	2	1
MAGNITUDE	3	2	1
SCORE	7-9	4-6	1-3
LOADING POTENTIAL	HIGH	MEDIUM	LOW

There is a fairly extensive literature base regarding various types of landuses, crops and agricultural practices, and the associated nitrate loading rates (Palone & Todd 1997). The area of each landuse in the management zone can be calculated from the orthophotograph or map, and the nitrate loading rate per unit area can be estimated. By summing and averaging the loading rates per unit area for each landuse, an average nitrate loading rate for the management zone can be estimated. Loading rates vary in response to fertilizer applications. Most agricultural crop products require fertilization, and fertilization schedules can be acquired for the farm or agricultural area. These schedules can help more accurately predict the loading rate in each management zone.

The three factors used in this example may give a better indication to the potential nitrate loading to a management zone. The greater the source density, intensity and magnitude of the various landuses, the greater the potential nitrate loading rate per unit area. High loading potentials within management zones may indicate a greater priority for buffer zone establishment, whereas loading potentials of lower ratings may indicate that buffer zone establishment for those management zones are of less priority.

Step 1c) To identify the delivery pathways within each management zone

The delivery pathway of water is very important when trying to prioritise management zones, as it can greatly influence a riparian buffer zone's ability to attenuate nitrate. It has been established in Chapter 3 that when the delivery pathway of water passes through the root zone of the riparian buffer zone, soluble nitrate is made available for vegetative uptake and denitrification. However when the delivery pathway of water bypasses this zone, nitrate removal is negligible. It is therefore important to determine the local geology of the micro-catchment. If the water follows a deep flowpath, bypassing the root zone of the riparian area, then a proposed riparian buffer zone will be ineffective at nitrate attenuation, and the management zone should no longer be considered for this type of nitrate management. If, however, the local geology promotes shallow flowpaths, then the management zone may remain a potential priority for riparian buffer zone establishment.

Step 1d) To order management zones according to priority

The management zones should be ordered according to priority, where highest nitrate loading receives highest priority if the delivery pathway of water for that management zone brings nitrate into contact with the riparian area. Other factors like proximity of natural wetland systems, other riparian corridors or important habitats in terms of terrestrial or aquatic conservation should also be noted. Management zones affected by these should be given higher priority. The riparian management zones are established and prioritised. The highest priority areas should then be assessed for their existing potential to remove nitrate. The following step in these guidelines is an example of how to achieve this.

STEP 2. TO ASSESS NITRATE ATTENUATION POTENTIAL IN EACH MANAGEMENT ZONE

With an understanding of the influential factors in nitrate attenuation in riparian buffer zones, it is possible to assess a specific site's potential for this function. The various characteristics of these influential factors will indicate how well vegetative uptake and denitrification are functioning at present. The basic site potential for nitrate attenuation is assessed according to its current soil and vegetation characteristics. The nitrate attenuation potential will give an indication of what management practices may be necessary for the site. Once the site potential for nitrate attenuation is established, a buffer zone of suitable vegetation can be created on the site.

Assessing the site potential for nitrate attenuation requires that the current vegetation is rated. This will give a better indication of the soil characteristics at present. Once the site potential for nitrate attenuation is achieved, a buffer zone can be established with appropriate vegetation (preferably a mix of indigenous species).

The guidelines focus on each of these two key mechanisms of nitrate attenuation. Once the management zones have been prioritised, each zone can be rated using the following assessment. The assessment requires that the user rate various influential factors, which are then scored and summed. The eventual score will indicate to the user the potential the site has for nitrate attenuation, and will also give a better idea of whether a buffer zone will be of benefit to the management aim. There are two parts to the assessment, each part focusses on one of the two key mechanisms of nitrate attenuation in riparian buffer zones. Each of the two parts are reduced to a score out of five. The two parts are summed at the end to give an overall estimate of a site's potential to attenuate nitrate out of ten.

$$\mathbf{NAP = DP + VUP} \qquad (1)$$

where:

NAP = Nitrate attenuation potential

DP = Denitrification potential

VP = Vegetative uptake potential

Step 2a). To establish the potential for denitrification

The process of denitrification requires three things (Chapter 3):

1. That nitrate be available to be utilised by microbial populations in the process of respiration
2. That the surface soils be anaerobic for most of the time such that oxygen is absent, and

nitrate is utilised instead by the microbial populations as an electron acceptor in the process of respiration

3. That organic carbon is available to be used as an energy source by the microbial populations in the process of denitrification.

As nitrate attenuation is the main management objective of this mini-dissertation, the first requirement for denitrification does not need to be assessed. The nitrate loading of a management zone is assessed in ***Step 1b***, and will contribute towards prioritising the management zone for buffer zone establishment for nitrate attenuation. As such, if the management zone has reached this part of the assessment, then it is assumed that the zone is a priority, and nitrate is therefore available for denitrification to occur. The second two requirements are used to determine a site's potential for denitrification. It is assumed that they are of equal importance and are therefore rated equally. To comply with the rating format in ***Step 2b***, the score for each requirement is halved.

$$\mathbf{DP = (SI \times 0.5) + (OMC \times 0.5)} \qquad (2)$$

where:

DP = Denitrification potential

SI = Saturation index (Table 4.1)

OMR = Organic matter content (Table 4.2)

The second requirement for the process of denitrification to occur (anaerobic soils), can be determined on a site by the saturation index of the soil (Table 4.3). If the soils are permanently saturated, then the process of decay is hindered and there will be little available organic carbon for the process of denitrification. If the soils are seldom saturated then there will be too much oxygen available for microbial respiration. For this reason, soils that are almost always saturated provide the optimum environment for denitrification.

Table 4.3: Scoring the saturation index of the soils

SATURATION INDEX (SI)				
ALMOST ALWAYS	USUALLY	SELDOM	ALWAYS	NEVER
5	**4**	**3**	**2**	**1**

The availability of carbon is the third requirement for the process of denitrification (Table 4.4). Organic carbon is made available through the process of decay, and therefore a high organic matter content will provide the soil with available carbon. The higher the organic matter content, the greater the potential for denitrification, and the better the score.

Table 4.4: Scoring the organic matter content of the soils

ORGANIC MATTER CONTENT (OMC)				
VERY HIGH	HIGH	MEDIUM	LOW	VERY LOW
>30 %	15-30 %	7-14 %	3-6 %	<3 %
5	**4**	**3**	**2**	**1**

Step 2b). To establish the potential for vegetative uptake

Chapter 3 outlined the process of vegetative uptake, and those factors enhancing the process. Three factors were important in the ability of plants to trap and absorb nitrate as a solute. These factors included:

1. the permeability of the soil
2. the delivery pathway of water
3. the existing vegetation structure

These factors can be rated and scored in the same way as those in ***Step 2a***. The scores are each weighted, and then summed to give a final rating of the potential of the site for vegetative uptake. It is assumed that each of the three factors are of equal importance and they are therefore weighted accordingly.

$$\mathbf{VUP = (SP \times 0.33^{\cdot}) + (WDP \times 0.33^{\cdot}) + (VS \times 0.33^{\cdot})} \qquad (3)$$

where:

VUP = Vegetative uptake potential

SP = Soil permeability (Equation 4)

WDP = Water delivery pathway (Equation 5)

VS = Vegetation structure (Table 4.11)

Soil permeablity:

The ease at which water moves through the soil will influence the water retention time, and the nitrate availability for vegetative uptake. Two factors have been chosen to help decide how easily water will be retained in the soil, namely: the runoff potential of the soil and the slope. It is assumed that the runoff potential and the surface gradient are of equal importance in influencing the ease of water movement through the soil, so the scores are summed and halved to comply with the rating systems of each of the other factors.

$$SP = (RP \times 0.5) + (S \times 0.5) \qquad (4)$$

where:

SP = Soil permeability

RP = Runoff potential

S = Slope

The runoff potential of the soil will indicate the potential for the soil to allow for infiltration, or its permeability. The inherent capacity of soils devoid of vegetation to allow for infiltration can be estimated according to the hydrologic soil group to which the soil belongs. A hydrologic soil group refers to soils grouped according to their runoff-producing characteristics (Palone & Todd 1997), and will therefore give an indication of infiltration potential. It is the saturated hydraulic conductivity and the internal free water occurrence of the hydrologic soil groups that influences infiltration (Table 4.5).

Table 4.5: The characteristics of the hydrologic soil groups relevant to this mini-dissertation

HYDROLOGIC SOIL GROUP	INTERNAL FREE H_2O OCCURRENCE	SATURATED HYDRAULIC CONDUCTIVITY	INHERENT RUNOFF POTENTIAL
GROUP A	Very deep	Very high	Low
GROUP B	Deep	High	Average
GROUP C	Shallow	Moderately high	Medium
GROUP D	Very shallow	Moderately low	High

The infiltration rate of the soils when thoroughly wet decreases from Group A to Group D. Soils in Group A are mainly deep, well drained and sandy or gravelly. Water moves very easily through these soils, and runoff potential is low. The water infiltrates to the deeper layers more easily as the depth to the water table is greater than two metres (Palone & Todd 1997). Water and solute nitrate is not retained in the biologically active layer of these soils, or the riparian root zone for very long. Soils in Group D tend to have a claypan or clay layer at or near the surface, with permanently high water tables (less than 60cm), or they are very shallow over nearly impervious bedrock (Palone & Todd 1997). Subsurface water is forced to remain as shallow flow, and is retained within the biologically active layer of the soil and the riparian root zone for longer periods of time. But the runoff potential of these soils is high, such that surface flow may not infiltrate the soils at all.

The soils of Group's B and C will allow water to infiltrate the soils (their runoff potentials are average and medium respectively), but the water tables in these soils are shallow (between 0.6 metres and 1.2 metres). The average to medium infiltration rates will promote solute nitrate trapping from surface flows, and water will remain shallow making vegetative uptake more possible. Therefore these soils receive a good score (Table 4.6), as their inherent permeability is more favourable for vegetative uptake than those of Groups A and D.

Table 4.6: Scoring the runoff potential of the site

RUNOFF POTENTIAL				
	GROUP C	GROUP B	GROUP D	GROUP A
SCORE	**5**	**4**	**3**	**2**

The potential for infiltration is further influenced by the gradient. A slope rating (Table 4.7) is added to the hydrologic soil group rating to include this influence. The surface flow velocity increases with steeper slopes, and reduces the possibility of infiltration. The retention time of the water in a riparian area is also lessened with an increase in slope. A slope greater than 15% is considered moderately steep with an average water retention time, and therefore of average nitrate attenuation enhancing abilities (Castelle *et al.*1994; Palone & Todd 1998; Klapproth 1999). The ratings and score were set accordingly.

Table 4.7: Scoring the general slope of the site

SLOPE					
	SLIGHT	**GENTLE**	**MODERATE**	**STEEP**	**VERY STEEP**
PERCENTAGE	0-7	8-15	16-25	26-35	>35
VELOCITY	VERY SLOW	SLOW	MODERATE	FAST	VERY FAST
SCORE	**5**	**4**	**3**	**2**	**1**

Water delivery pathway:

The delivery pathway of water will most definitely influence a riparian buffer zone's ability to attenuate nitrate (Section 3.4.1). ***Step 1c*** determined whether water bypasses the riparian area or not. Only catchments where delivery pathways (both surface and subsurface) pass through the riparian zone are considered here. The assessment rates both flow types, which are each given an equal rating as it is assumed that both are equally influential in nitrate retention by vegetative uptake. The scores for each are summed and halved to comply with the rest of the assessment.

$$\textbf{WDP} = (\textbf{SSF} \times 0.5) + (\textbf{SF} \times 0.5) \qquad (5)$$

where:

WDP = Water delivery pathway

SSF = Subsurface flow (Table 4.8)

SF = Surface flow (Table 4.9)

Subsurface flows (SSF) are often retained in riparian buffer zones because of the reduced surface gradient at the river level. Subsurface flow following shallow lateral delivery pathways will be easily reached by root systems for uptake by plants (Section 3.2.1), and nitrate will be utilised by microbial populations in the biologically active layers of the soil (Section 3.4.1). RBZ's in this environment would be most effective at attenuating nitrate, and micro-catchments with this geology will receive a score of 5 (Table 4.8). However subsurface flows following deeper delivery pathways may bypass root systems of grass species and the biologically active layers of the soil. RBZ's established in micro-catchments with geologies promoting deeper delivery pathways would only be effective at attenuating nitrates if planted with deep rooting trees. This flow type receives a score of 1.

Table 4.8: Scoring the subsurface flow types of the site

SUBSURFACE FLOW TYPES					
DEPTH	V. SHALLOW	SHALLOW	MEDIUM	DEEP	VERY DEEP
cm	<10	11 - 20	21 - 40	41 - 80	>80
SCORE	5	4	3	2	1

Surface (SF) run-off may be both channelled or dispersed. Vegetation will be most influential in trapping dispersed surface run-off, and promoting its infiltration, where it will then be made available for plant uptake or denitrification (Chapter 3). Surface runoff travelling as sheet flow receives a good score (Table 4.9). Nitrate in channelled surface flow is less easily trapped by riparian vegetation, and receives a lower score.

Table 4.9: Scoring the surface flow types of the site

SURFACE FLOW RATING					
	SHEETFLOW	DISPERSED	RILLING	GULLYING	CANALS
SCORE	**5**	**4**	**3**	**1**	**0**

Vegetation structure:

The structure of the vegetation will influence how easily the delivery flow path of water is reached, and how well soluble nitrate is trapped or utilised by the plants. Chapter 3 concluded that the type of vegetation was not necessarily as influential in nitrate attenuation as vegetation structure. Four factors influence vegetation structure. These four factors are rated (Table 4.10) and scored (Table 4.11).

Plants of varied root systems and dense stem densities, that provide good groundcover provide the optimum potential for nitrate attenuation in plant uptake. Plants of varied ages, that frequently provided leaf litter to the soils for improved organic matter contents and available carbon, will promote the process of denitrification.

Table 4.10: Rating the vegetation structure

VEGETATION STRUCTURE					
	VERY HIGH	HIGH	MEDIUM	LOW	VERY LOW
1. ROOTING DEPTH	5	4	3	2	1
2. ROOTING DENSITY	5	4	3	2	1
3. STEM DENSITY/COVER	5	4	3	2	1
4. LEAFLITTER POTENTIAL	5	4	3	2	1

The sum of the four factors will give a rating out of a maximum of 20. This rating needs to be scored (Table 4.11) as the other factors have out of 5, so as not to affect the weighting. The final rating is shown below:

Table 4.11: Scoring the vegetation structure

VEGETATION STRUCTURE RATING					
SCORE	17-20	13-16	9-12	5-8	4
RATING	**5**	**4**	**3**	**2**	**1**

Step 2c). To score the nitrate attenuation potential

The assessment requires that ***Step 2a***, which is a site potential rating for denitrification out of a possible score of five, be added to ***Step 2b***, which is a site potential for nitrate attenuation by vegetative uptake out of a possible score of five. The total score out of ten will give an indication of the given site potential to attenuate nitrate (Table 4.12).

Table 4.12: Scoring an estimate of nitrate attenuation potential

NAP = DP + VUP (1)					
DP + VUP	8 -10	6 - 8	4 - 6	2 -4	1 - 2
POTENTIAL	VERY GOOD	GOOD	AVERAGE	POOR	VERY POOR
EFFICIENCY	>70%	56 - 70%	46 - 55%	36 - 45%	<35%

where:

NAP = Nitrate attenuation potential

DP = Denitrification potential

VP = Vegetative uptake potential

If a management zone has an average to very poor potential for nitrate attenuation, then it is more important that a riparian buffer zone is designed for that management zone. Management zones receiving higher ratings are functioning fairly well at nitrate attenuation, and designing and establishing riparian buffer zones for these management zones is of less priority.

4.3.1 Establishing a riparian buffer zone

There are many existing guidelines on how to establish riparian vegetation for both water quality and habitat functions (Cooper *et al.* 1986; Budd *et al.* 1987; Cohen *et al.* 1987; Osborne & Kovacic 1993; Barling & Moore 1994; Leeds-Harrison *et al.* 1996; Dickson & Shaeffer 1997; Karssies & Prosser 1999; Lowrance *et al.* 2000a). These guidelines suggest that indigenous plant species are used, and describe the best planting methods and management techniques. Many compare grassed or forested systems, and recommend one type for a specific function. For the function of nitrate attenuation, both grassed and forested buffers are similarly effective, and it appears that there is little influence of type of vegetation in nitrate attenuation (Section 3.4.2). It is acknowledged that these guidelines are not extensively tested in South Africa.

The role of riparian buffer zones in nitrate attenuation is more affected by the structure and density of the vegetation. It is recommended that the vegetation be of varied age and height, stem structure, root density and depth, with mixed deciduous and evergreen plant species.

Mixed vegetation of varied stem structure will trap surface runoff well and promote infiltration, whilst varied root systems will reach various levels of subsurface flow. Deciduous plant species will supply leaf litter to the soil for denitrification, whilst evergreen plant species will provide year-round canopy cover, shade and habitat. Indigenous vegetation will establish well on its own, and is usually the best option for riparian areas. As the topic of buffer zone establishment and management is well covered by world literature, it will not be dealt with in detail in this mini-dissertation. Instead, the problem of deciding on adequate buffer size specifically for nitrate attenuation will be discussed in the next section. The guidelines follow on from ***Step 2*** above, and are an example of how the linear width of a riparian buffer zone may be decided. The linear nature of streams and rivers most often require a linear buffer, and for this reason an average linear buffer width may be established for a management zone, using these guidelines.

4.4 DETERMINING RIPARIAN BUFFER WIDTH

Possibly one of the most controversial and debated aspects of riparian rehabilitation and design is the determination of adequate buffer width for the desired riparian function. The size or width of a riparian buffer strip greatly affects the effectiveness and sustainability of the buffer (Palone & Todd 1997). Oversized buffers may deny landowners a portion of land, whereas undersized buffers may be ineffective and reduce the value and sustainability of the aquatic resource (Castelle *et al.* 1994). These situations may be economically or environmentally costly, urging consensus on how to determine the necessary minimum buffer width for a given riparian habitat.

The many approaches and formulas developed world-wide regarding buffer width may be based on specific site studies, desired riparian functions, adjacent land uses, legal, socio-economic and political decisions, or even on the specific field of interest of the researcher concerned. The variety of criteria used has led to an inconsistency in recommendations and disagreement amongst riparian researchers and managers world-wide.

For example, in the Pacific Northwest it is recommended that a riparian buffer be equal to the maximum possible tree height for that site, for each side of the stream. Riparian buffer widths range from 11 metres to 38 metres (Budd *et al.* 1987). Yet it has been suggested by a hydrologist in Minnesota, USA that the proper riparian width for effective management is the inclusion of the active 50-year flood plain and the terrace slopes, which can be calculated as ten times the stream 'bankfull' width plus 15 metres on either side (Klapproth 1999). In Chesapeake Bay, USA, it is recommended that the minimum width for riparian buffers is specific to desired function, where 10 metres on each side of the stream is deemed sufficient

for benefits to aquatic communities, with the buffer increasing to 23-30 metres for the improvement of water quality amelioration and habitat functions (Palone & Todd 1997). Riparian Buffer zones in Central Pennsylvania, USA are usually 30 metres in width on either side of the perennial and intermittent
streams (Lynch *et al.* 1985).

Swedish authors, Petersen *et al.* (1992) recommend that riparian buffers should be at least 10 metres in width on either side of the stream, but should be based on nutrient retention and habitat considerations of the stream in question. There are no clear guidelines, however, as to how these two considerations should influence buffer width determination.

Canada and Australia base many of the buffer width recommendations on observations made pertaining to sediment travel through different vegetation types. For example, it is generally recommended that a riparian buffer in a forested area needs to be a minimum of 30 metres for maximum sediment retention efficiency, but may be dependant on site specific conditions.

In Queensland, Australia, grassed filter strip widths are also determined by the nature of sediment travel. Riparian buffer width determination for surface flow (Karssies & Prosser 1999) is dependent on:

1. The amount of sediment to be trapped (or the soil loss rates for the site).
2. The particle size of the suspended sediment.
3. The shape of the immediate drainage catchment.

The sediment storage capacity of a riparian filter strip will increase with width (Karssies & Prosser 1999), and therefore the more sediment the filter is required to trap, the wider the filter strip will need to be. If the adjacent landuse practices good on-site conservation measures for the prevention of erosion and sedimentation, then the buffer width can be decreased as the amount of sediment to be trapped is considerably less. Fine particle-sized sediment will require a wider buffer strip, as fine suspended sediment is able to move further into the filter strip before settling out of suspension. The shape of the drainage catchment will influence the degree of convergence, and thus the amount of suspended sediment traveling in concentrated flow (Karssies & Prosser 1999). Concentrated flow has a greater velocity and load capacity, and will carry the suspended sediment further into the buffer zone. Areas of greater convergence will require buffers of increased width (Figure 4.1).

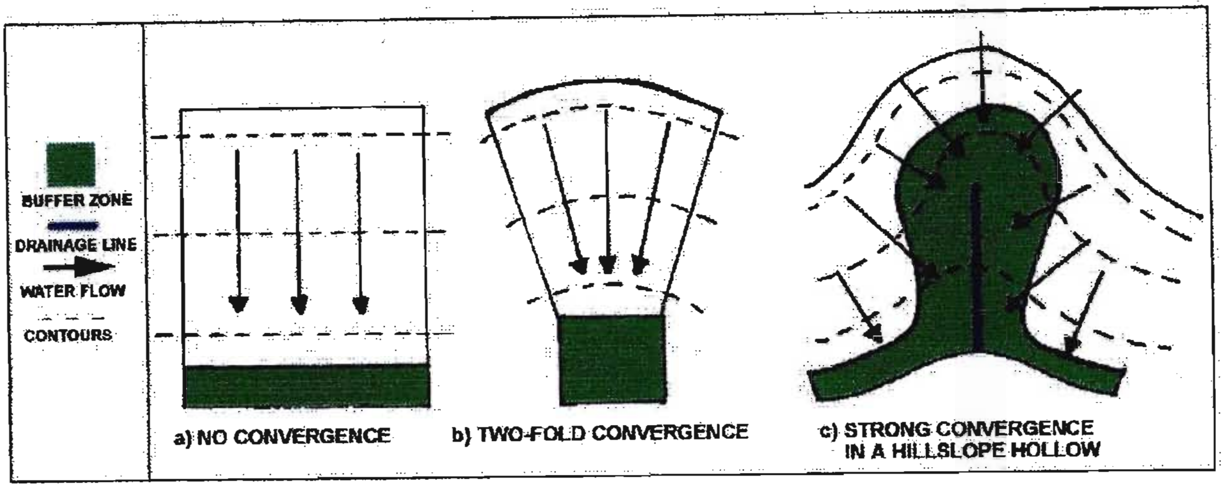


Figure 4.1: The influence of drainage catchment shape on riparian width determination (after Karssies & Prosser 1999)

Neither of the first two criteria for buffer width determination used in Queensland, Australia (Karssies & Prosser 1999) are particularly useful if the desired riparian function is nutrient retention. The focus of these two criteria is on the surface-sediment trapping abilities of buffer zones, and the function of soluble nutrient attenuation is neglected. Nitrate as a solute is often more obvious in groundwater than in surface flow (Prosser *et al.* 1999b). Therefore these criteria are of little use when the management aim is nitrate attenuation. The third criterion as listed by Karssies and Prosser (1999) may be of some use as it considers the route of nitrate entry, although the focus is again the travel of sediment. The route of soluble nitrate entry in a buffer zone will influence the removal abilities of the buffer (Chapter 3). So consideration of the shape of the catchment will be of benefit to determining a suitable buffer width for nitrate attenuation.

In Victoria, Australia, the following formula is used to determine the width of a buffer strip:

$$W = 8 + 0.6\,S$$

where W is the buffer strip width (m), and S is the slope (%) (Barling & Moore 1994). Slope is a useful indicator of the rate of water movement and the degree of infiltration (Chapter 3). But when considering nitrate attenuation, other factors are also influential in the buffer's ability to retain nitrate. The type of soil present, or the nature of the vegetation may alter the influence the slope may have on water movement. This equation for buffer width determination may then fall short when nitrate attenuation is the main management aim.

Other riparian managers, decision makers or regulatory agencies make use of stream, wetland and adjacent land use rating systems to determine suitable minimum buffer widths. Aquatic resources achieving higher scores (according to the chosen rating system) will be given priority in terms of management or rehabilitation and are afforded larger riparian buffer habitat, as opposed to narrower buffer widths for aquatic resources achieving lower scores. However, these rating systems fail to incorporate many criteria affecting riparian habitat function as identified by much of the literature (Castelle *et al.* 1994).

The diversity of environments, climates, watersheds, land uses and desired functions of riparian buffer zones make any one of the aforementioned formulas or recommendations regarding buffer width insufficient to be adopted more widely, especially when nitrate attenuation is the main management aim. Instead it is necessary to incorporate all criteria associated with buffer width determination identified by the available literature.

4.4.1 Using scientifically based criteria

Palone and Todd (1997) suggest that four important scientifically based criteria (Table 4.13) should be considered when determining adequate buffer width. The determined width can then be modified by socio-economic variables such as management objectives or constraints. The authors do not indicate specifically how these four criteria should be used, but rather discuss how each criterion affects riparian buffer function. It is argued that buffer width should increase to a decided maximum, or decrease to a decided minimum according to the on-site criteria. The authors give no indication of how to decide the maximum or minimum width, however.

Table 4.13: Criteria for the determination of riparian buffer width as identified by Palone &Todd (1997)

CRITERION 1 Existing or potential value of the resource to be protected
CRITERION 2 Specific water quality and/or habitat functions desired
CRITERION 3 Site, watershed and buffer characteristics
CRITERION 4 Intensity of adjacent landuse

i) Criterion 1: Value of the resource

Palone and Todd (1997) indicate that a resource (wetland, stream, lake or shorezone) of higher functional value will require a wider riparian buffer than that of lower functional value. It is difficult to determine 'value' as it involves subjective judgement, but the authors suggest that scientific evaluation can be used to establish this. It is further suggested that watershed, water quality and habitat conditions may be useful as criteria for determining the necessary buffer width for the resource. Degraded watershed, water quality or habitat conditions in the authors opinion, would require a wider riparian buffer zone. This would be dependant however, on the value of the given water resource such that a highly valued water resource should be afforded a wider buffer zone (Figure 4.2). It is perhaps more useful to afford priority to the less degraded resources however, as these are more easily rescued. The key elements necessary for rehabilitation would be more easy to attain in a less degraded environment, and chances of success would be greater.

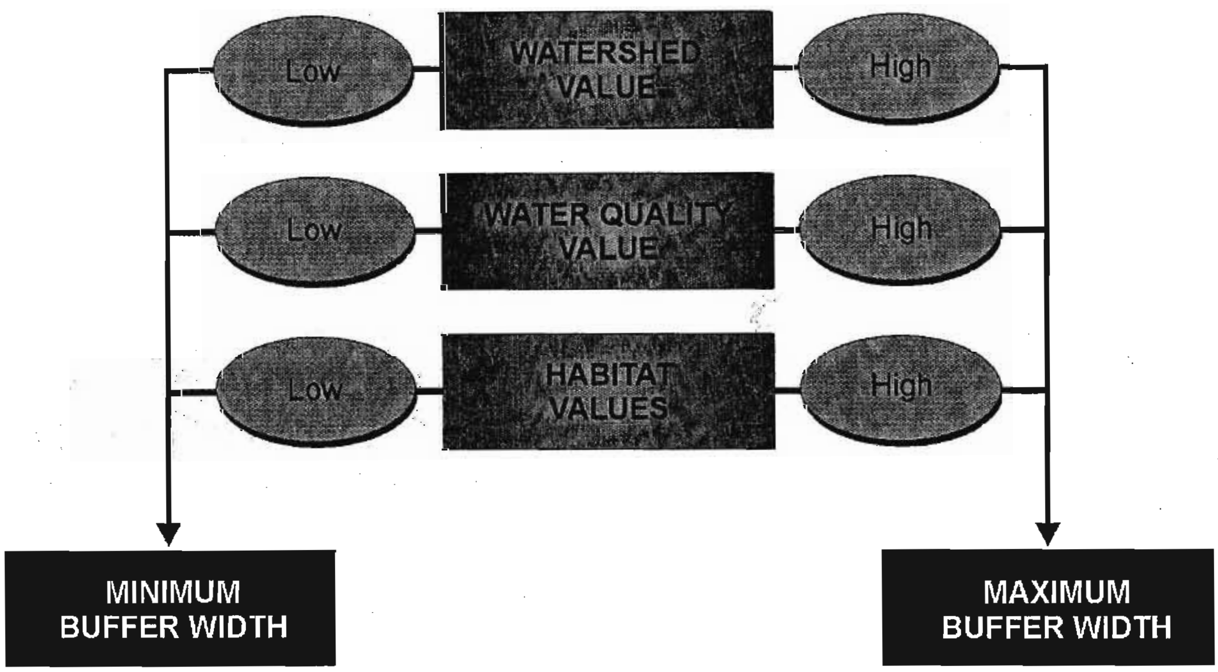


Figure 4.2: The influence of criteria on the value of the water resource when determining buffer width

ii) Criterion 2: Desired buffer function

It is deemed important by Palone and Todd (1997) to evaluate the range of buffer functions required under site-specific conditions, before the necessary buffer width can be determined. The buffer widths necessary for the range of buffer functions to be successful vary widely. Palone and Todd (1997) have established very generalised minimum buffer widths required for specific buffer functions (Figure 4.3).

The authors suggest a minimum buffer width of 9 to 39 metres for the efficient buffer function of nitrogen removal. They acknowledge the influence the pathway of water may have on the function of nitrogen removal, and therefore offer a range of 30 metres in buffer width. Figure 4.3 gives a generalised idea of the buffer width necessary for the combination of particular functions identified. But it is still difficult to estimate a more accurate, cost effective and ecologically successful riparian buffer width from this system.

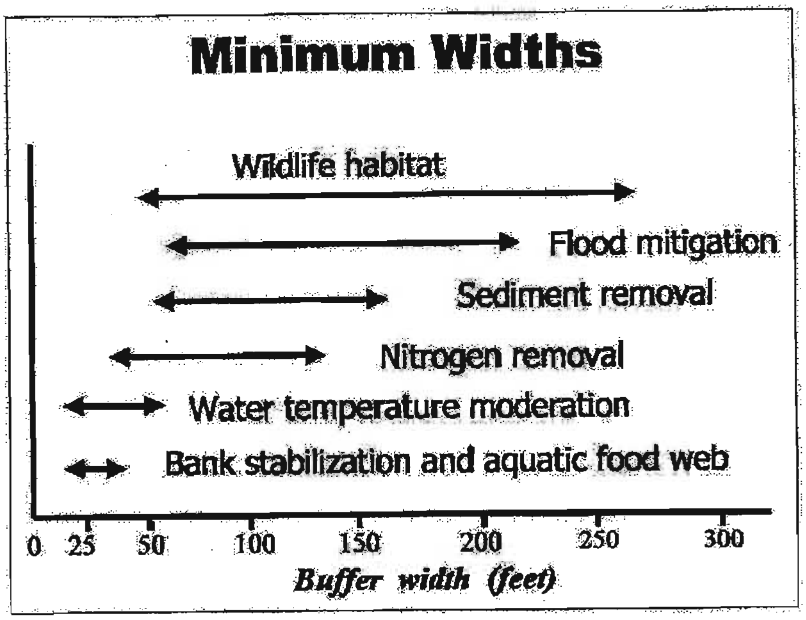


Figure 4.3: Range of minimum buffer widths for meeting specific buffer objectives (as cited in Palone & Todd 1997)

iii) Criterion 3: Site, buffer and watershed traits

The determination of buffer width is particularly dependant on site factors when pollutant removal is the most desired buffer function, as this function is more specifically affected by individual site differences. Palone and Todd (1997) consider a range of site criteria for buffer width which influence the functions of a riparian buffer (Table 4.14). The determination of the exact minimum or maximum buffer width should include criteria where environmental sustainability is not jeopardised, and economical cost is considered. In this way, important agricultural land is not unnecessarily lost to riparian zoning, and ecological functions are not compromised by too little riparian environment.

This mini-dissertation is particularly concerned with the removal of nitrate by a riparian buffer system, and as such will consider the site criteria most influential in the ability of a riparian environment to fulfil this function. These specific site criteria can be divided into three broad groups, namely: hydrogeology, vegetation, and soils.

The hydrogeological criteria should be considered collectively to indicate the width at which the buffer should be designed. The route and rate of water movement, depth to the water table, surface and soil slope gradients, and the geology of a site are all influential hydrogeological factors. The way in which they influence buffer width determination is depicted in Figure 4.4. The discussion regarding hydrogeology in Section 3 .4.1 concludes that when the route of water movement follows a shallow lateral flow path, nitrate removal by the riparian zone is maximised. This is because the soluble nitrates in the subsurface flow are easily reached and absorbed by the root systems of the riparian vegetation. The microbial processes involved in nitrate retention and removal are also more active in the surface layers of the soil. The gradient of the surface and soil slopes in the riparian environment will influence both the rate and route of water movement. A steeper gradient will increase the flow velocity, reducing the water infiltration or retention times within the zone. Soil microbial processes and vegetative uptake of soluble nitrate is therefore reduced. It follows that very steep gradients will require the maximum buffer width afforded to the site in question. The depth to the water table influences the soil microbial processes, and the availability of water for vegetative uptake. A water table depth greater than 38cm will require a very wide buffer zone, as the nitrate removal processes are reduced by it's depth.

Table 4.14: Site criteria identified by Palone & Todd (1997) important in buffer width determination

SITE CRITERIA AFFECTING BUFFER WIDTH
• Watershed condition • Slope • Stream Order • Soil depth and erodibility • Hydrology • Flood plains • Wetlands • Stream banks • Vegetation type • Stormwater system

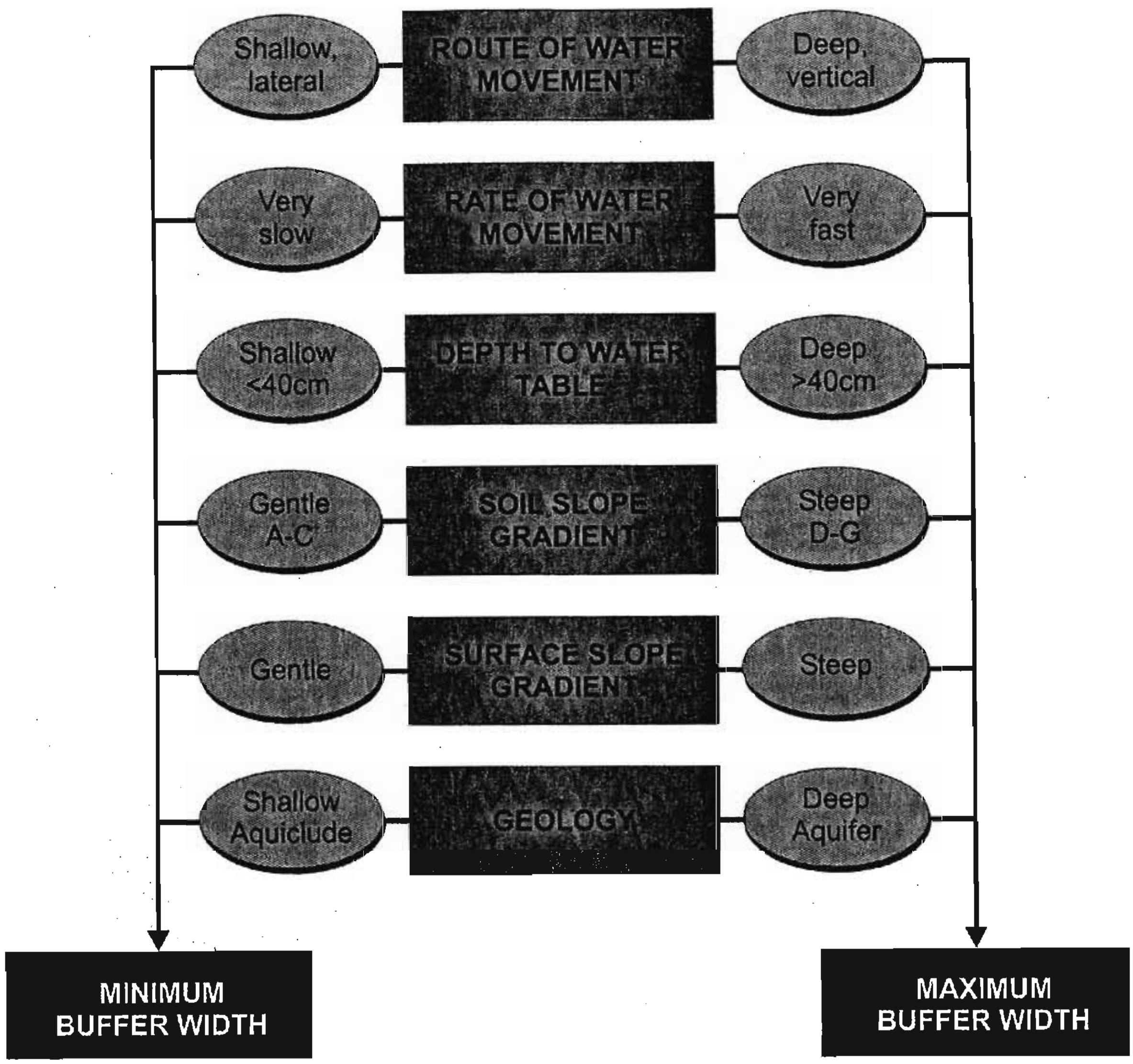


Figure 4.4: The influence of hydrogeological criteria on the determination of riparian buffer width

The influential factors related to vegetation include the condition of the buffer zone, the root depth and density, the stem density, the leaf litter potential, and the cover and shade percentages of the vegetation (Figure 4.5). If the riparian environment is poorly covered, or has low stem and root densities, it would then be necessary to increase the buffer zone to the maximum width afforded to the site. Factors of this status would reduce the trapping and filtering abilities of the riparian zone, and expose soil to greater erosion.

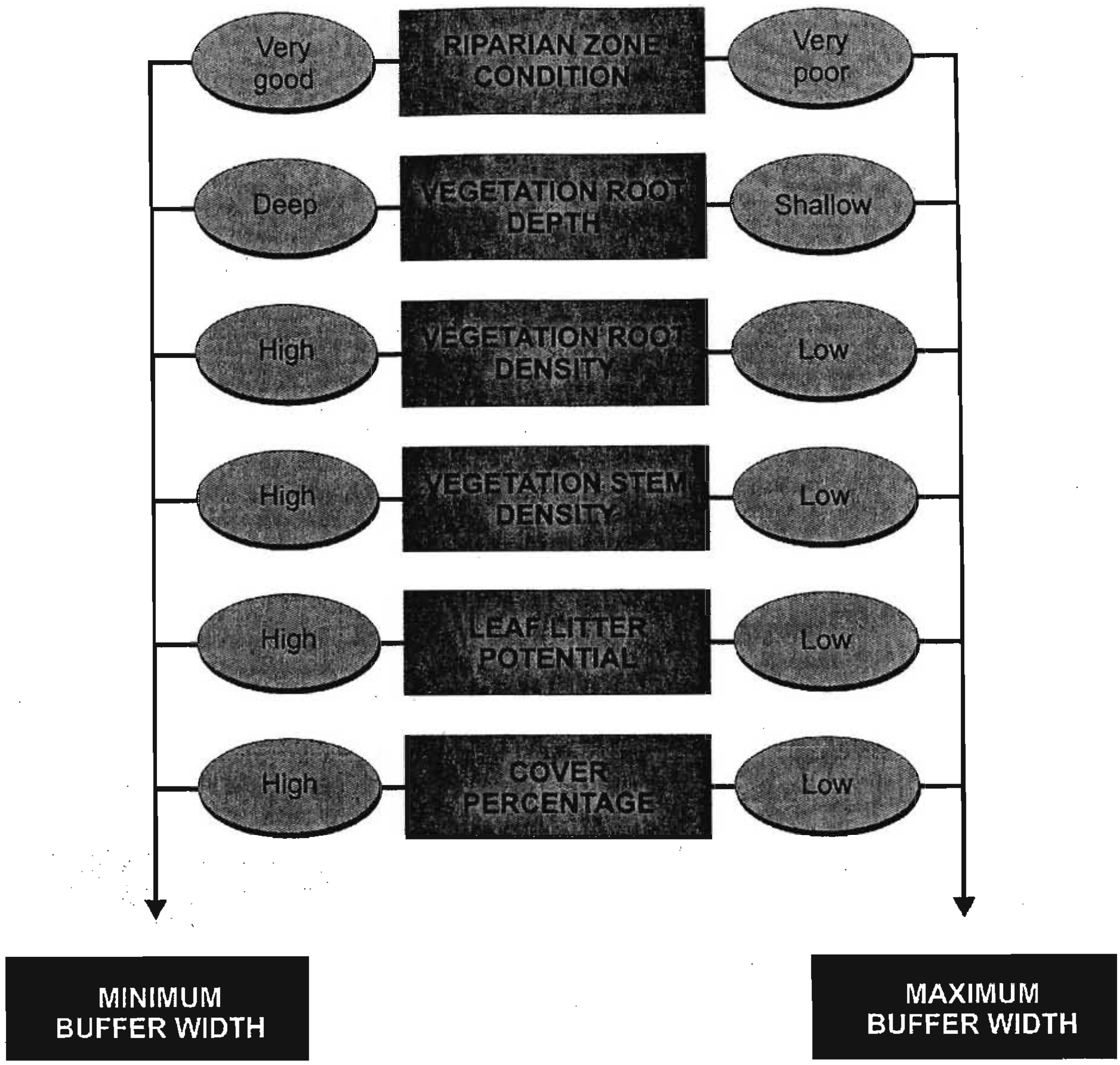


Figure 4.5: The influence of vegetation criteria on the determination of riparian buffer width

Poor leaf litter provides little friction to runoff, and insufficient cover to erosion, as well as reducing the site potential for available carbon and organic matter (factors necessary for the process of denitrification). It follows that a wider buffer zone would be necessary when these conditions are evident, providing a larger area for nitrate removal processes to occur.

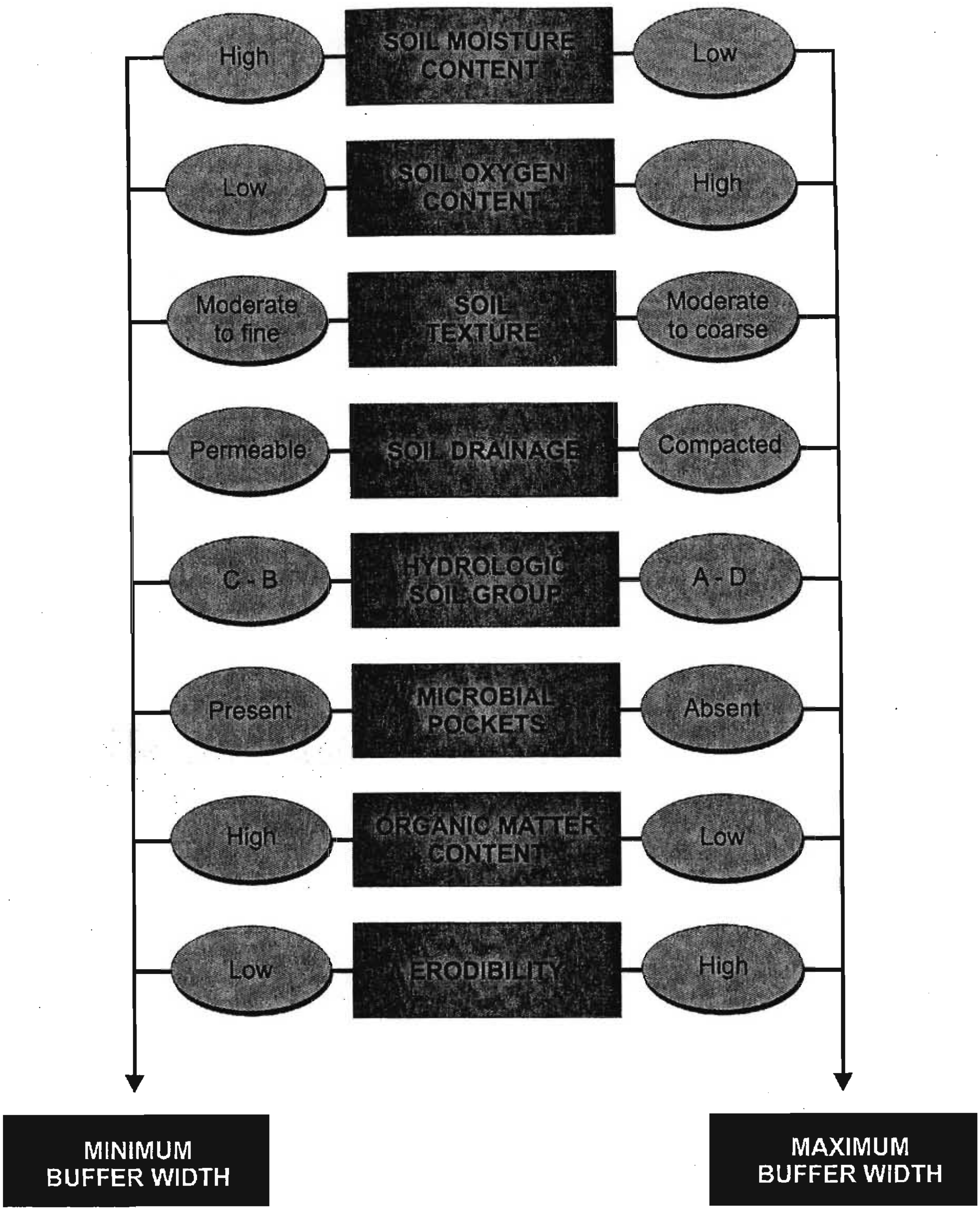


Figure 4.6: The influence of soil criteria on the determination of riparian buffer width

Soil characteristics most influential in nitrate removal have been discussed in Section 3.4.2. Soil moisture; oxygen and organic matter content, soil texture; drainage and erodibility, the hydrologic soil group and presence of microbial pockets all affect the nitrate removal potential of the riparian buffer zone (Figure 4.6). The soil moisture content, oxygen content, organic matter content and presence of microbial pockets will determine the potential for the process of denitrification to occur. The less potential there is on a site for denitrification, the greater the buffer width will need to be. Soil drainage and the potential for erosion will affect the moisture content and the infiltration potential of the soil, and the retention time of the water and attached nitrates. Both vegetative uptake of nitrates and denitrification are influenced by these factors. A soil that is well-drained (permeable) with a low potential for erosion, will promote sediment and attached pollutant deposition, and would therefore require only a narrow buffer width. Moderate and fine-textured soils do however, promote the process of denitrification (Palone & Todd 1997) and would in theory require a narrower buffer zone too. But if the soils are too compacted or highly erodible, little water infiltration or sediment deposition would occur, and a wider buffer zone would be necessary.

iv) Criterion 4: Intensity of adjacent landuse

The negative effects of an activity are often buffered from adjacent landuses of different activities. For instance an airport may be buffered from residential developments, with the size of the buffer proportional to the impact of the negative effects of the activity. Similarly the impact of adjacent landuses on the adjacent riverine ecosystems will require a buffer zone of suitable proportions (Figure 4.7). The density, intensity and magnitude are considered by Palone and Todd (1997) as useful indicators of the potential impacts of adjacent landuses to aquatic environments. As such, if the adjacent landuse density and intensity is high, and the magnitude of the landuse is large, then the maximum decided buffer width would be required. The potential nitrate yield of the adjacent landuse should also be considered when nitrate removal by the riparian buffer zone is required. The higher the nitrate yield of the adjacent activity, the greater the buffer width will need to be.

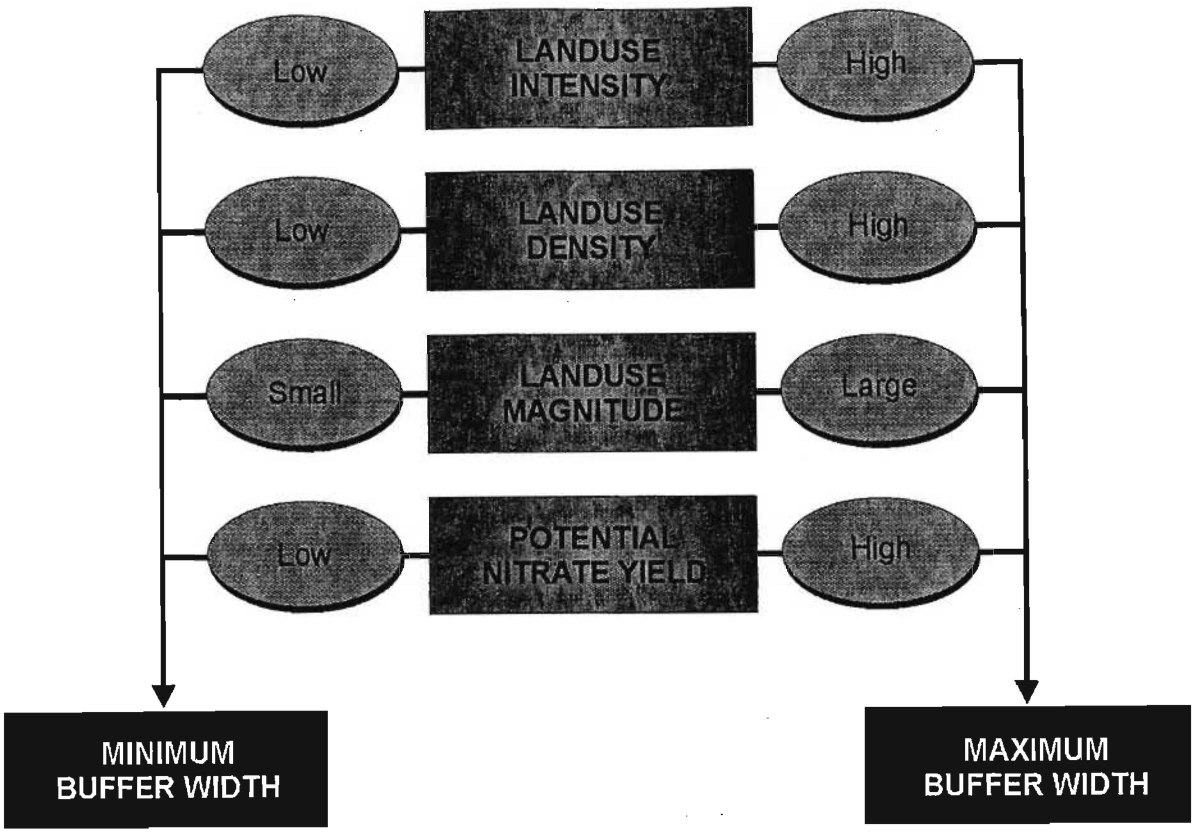


Figure 4.7: The influence of adjacent landuses on the determination of riparian buffer width

4.4.2 Developing criteria specific to nitrate removal

As previously mentioned, Palone and Todd (1997) indicate the influence the four criteria they describe will have on a buffer system, and are therefore able to comment on how the criteria influence buffer width determination. Yet the authors do not indicate how the user is to integrate the four criteria to determine a cost effective and economically sustainable buffer zone of practical width. Furthermore, it is difficult to accurately determine from this system, a buffer width specifically necessary for nitrate removal.

Many of the international recommendations on how to determine buffer width focus specifically on sediment and attached pollutants entering the riparian system in surface flow. But there is little consideration of width determination as based on the processes of nitrate removal, namely vegetative uptake and denitrification. Little attention has been afforded to trapping nitrate in subsurface flows. This mini-dissertation proposes that width determination should consider the site potential for promoting these two main processes. For these reasons the following guidelines have been developed to help determine a practical buffer width for the removal of nitrate specifically from subsurface flows for any given site, where important site-specific information for each criterion is scored.

STEP 3. TO DETERMINE ADEQUATE BUFFER WIDTH

The choice of vegetation for a riparian buffer zone will directly affect the interception of soluble nitrate in surface flow. The vegetation should therefore be of adequate stem and root structure and density to meet this objective. Thus buffer width determination through these guidelines will not consider the characteristics of soluble nitrate in surface flow. The following step will only rate those characteristics of the riparian zone which are influential in subsurface nitrate attenuation. Four influential factors in the subsurface nitrate removal ability of a riparian buffer zone are considered:

1. Organic matter content (as a percentage)
2. The hydrologic soil group
3. Slope or gradient of the surface soils (as a percentage)
4. The route of subsurface flow (in cm)

The four factors have already been considered in ***Step 2.*** These four directly relate to nitrate removal from subsurface flows. The rating scale for each factor remain the same as in ***Step 2***, but the scores will differ.

Each factor can be rated, and issued a score for that rating. The recent reviews on the subject of nitrogen attenuation (Castelle *et al.* 1994; Palone & Todd 1997; Dosskey *et al.* 2000) suggest that buffer zones narrower than 5 metres are ineffective at nitrogen attenuation (Figure 2.7). The widest recommended width according to these reviews for the best possible attenuation of nitrogen does not exceed 80 to 90 metres. Based on these findings, it was decided that a buffer width of 80 metres would be the maximum required buffer width for the function of nitrate removal to be effective in a riparian environment of poor nitrate removal potential. Similarly, it was decided that a buffer width of less than 5 metres would be ineffective at nitrate removal or unsustainable in the long term, even on a riparian site of good nitrate removal potential. As such, the scoring system ranged from 5 to 80. The scores are directly related to the width of the buffer zones, where higher scores will necessitate wider buffer zones. The rating category for each factor doubles in value, and as such, the scores double too.

Step 3a). Scoring the organic matter content

A soil is considered an organic soil if it contains an organic matter content (OMC) of more than 20% (Palone & Todd 1997). The percentage of organic matter in a soil is rated from very low to very high according to Palone and Todd's (1997) description above. The lower scores are given to the more organic soils, as organic matter is important for the process of denitrification, and their presence improve nitrate attenuation (Table 4.15).

Table 4.15: Scoring the organic matter content rating

ORGANIC MATTER CONTENT RATING (OMR)					
	VERY HIGH	HIGH	MEDIUM	LOW	VERY LOW
PERCENTAGE	>40 %	20-40 %	10-20 %	5-10 %	<5 %
SCORE	**5**	**10**	**20**	**40**	**80**

Step 3b). Scoring the hydrologic soil group rating

The characteristics of the hydrologic soil groups were established in ***Step 2b***. Soils are grouped according to their infiltration potential, drainage, level of the water table, and saturated hydraulic conductivity. Soils allowing for infiltration, but promoting shallow water retention are of optimum value to improving nitrate attenuation. Sites with soils of Group C therefore require a smaller buffer than those of Group A (Table 4.16).

Table 4.16: Scoring the hydrologic soil group

HYDROLOGIC SOIL GROUP RATING (HSGR)				
	GROUP C	GROUP B	GROUP D	GROUP A
SCORE	**10**	**20**	**40**	**80**

Step 3c). Scoring the slope rating

Both the surface and the subsurface gradients will influence the route and rate of water movement. As most of the site potential for nitrate removal is dependent on the biologically active zone of the soil (in the top few centimetres), it was decided that the subsurface gradient was not as important. For this reason 'slope rating' is simply the average surface gradient of the given site, and is rated from slight to very steep (Table 4.17). The water table generally follows the surface gradient of sloping land (Pinay & Burt 2001), which is also influential in nitrate attenuation.

Table 4.17: Scoring the slope rating

SLOPE RATING					
	SLIGHT	**GENTLE**	**MODERATE**	**STEEP**	**VERY STEEP**
PERCENTAGE	0-7	8-15	16-25	26-35	>35
VELOCITY	VERY SLOW	SLOW	MODERATE	FAST	VERY FAST
SCORE	**5**	**10**	**20**	**40**	**80**

Step 3d). Scoring route of subsurface flow rating

Subsurface water movement influences the nitrate availability for vegetative uptake and denitrification. Subsurface flow (SSF) is rated from *very shallow* to *very deep* (Table 4.18). The denitrification process is most impressive in the biologically active zone, which is usually in the top few centimetres of the soil. Therefore shallow SSF's are given a lower score, as this increases the nitrate removal function of the buffer zone.

Table 4.18: Scoring the subsurface flow rating

SUBSURFACE FLOW RATING					
DEPTH	V. SHALLOW	SHALLOW	MEDIUM	DEEP	VERY DEEP
cm	<10	11 - 20	21 - 40	41 - 80	>80
SCORE	5	10	20	40	80

Step 3e). Achieving a width estimate

It is assumed that each of the four factors are of equal importance to the nitrate removal ability of a riparian buffer zone and are given equal weighting (0.25), and the sum of the four weighted scores is a more accurate prediction of a suitable buffer width for the site:

$$\mathbf{TBW = (OMR \times 0.25) + (HSGR\ RATING \times 0.25) + (SR \times 0.25) + (SSFR \times 0.25)} \qquad (6)$$

where:

TBW = Total buffer width

OMR = Organic matter content rating (Table 4.15)

HSGR = Hydrologic soil group rating (Table 4.16)

SR = Slope rating (Table 4.17)

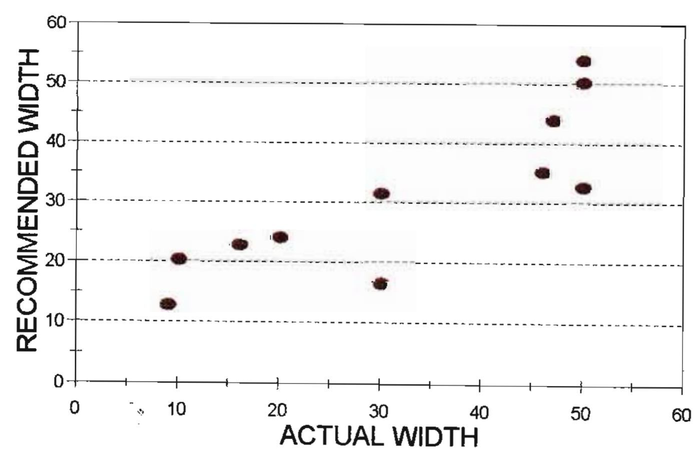
SSF = Subsurface flow rating (Table 4.18)

4.4.3 Test effectiveness

The proposed guidelines and rating system have been based on broad principles and assumptions in the absence of specific understanding. For this reason it was necessary to apply the rating system (to determine adequate buffer width for a given site) to the results of existing studies (Table 4.19). In this way it is possible to test the system by applying the guidelines to individual studies, and comparing the results to the actual findings of the individual studies. Studies that reported nitrate removal efficiencies of 65% to 100% have been used, as the buffer widths of these environments have proven extremely efficient at the task. The Pearson Correlation Coefficient is equal to 0.829, and significant at the 0.01 level.

Table 4.19: Test effectiveness as compared to studies of efficient nitrate removal, also displayed graphically

		RECOMMENDED WIDTH
Brusch & Nillson (1993)	20	23.75
Cooper *et al.* (1986)	16	22.5
Corley *et al.* (1999)	10	20
Dillaha *et al.* (1989)	9	12.5
Gambrell (1975)	46	35
Jacobs & Gilliam (1985a)	50	32.5
Jacobs & Gilliam (1985b)	47	43.75
Jordon *et al.* (1993)	30	16.25
Lowrance *et al.* (2000)	50	50
Peterjohn & Correll (1983)	50	53.75
Sloan *et al.* (1999)	30	31.25



4.5 THE CASE STUDY

Sourveld Farm is a potato, maize and dairy farm situated in the Kamberg-Kangatong area, west of Nottingham Road, KwaZulu-Natal. The Rosette/Giants Castle secondary road (D164) denotes the southern boundary of the farm, whilst the Klein Mooi River marks the northern boundary. The Sourveld Farm is approximately 450 hectares, and practices individual fertilizer applications for each of the three main agricultural activities.

The orthophotograph (1:10 000) 2929 BD 7 Kangatong was used to understand the drainage catchment on the Sourveld Farm (Figure 4.8). The small catchment drains part of the Kamberg mountain range and the Sourveld Farm, through a series of man-made dams, before entering the Klein Mooi River. From the orthophotograph it was evident that there are three small wetland systems throughout the farm's catchment.

Two main tributaries of the Kamberg watershed flow through wetland systems, before entering the Sourveld Farm under road culverts. The water then flows through the first set of wetlands (approximately 500m^2) on the Sourveld farm and into the first of two large water storage dams (top dam). The second wetland (Plate 4.1) is a small wetland (about 150m^2) filtering overflow water from the top dam before it enters the second large water storage dam (bottom dam). Overflow from the top dam follows a small grassed channel, and flows under the farm road along a concrete drain, where it enters the second wetland from an outlet pipe about 60cm in diametre. The outlet pipe is placed about one metre above the wetland system.

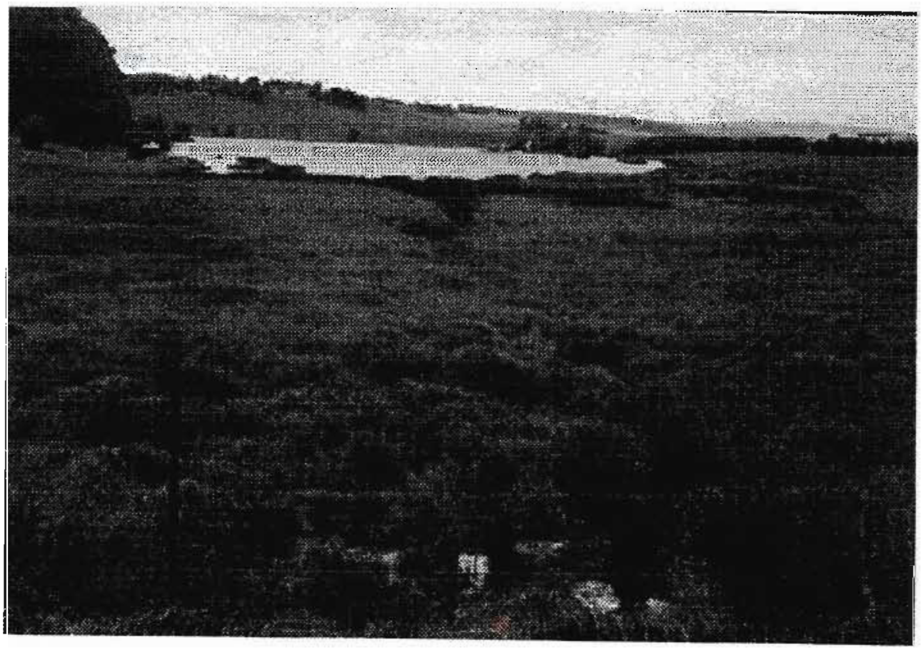


Plate 4.1: View from Top dam of wetland system above Bottom dam

Figure 4.8: The Sourveld Farm study site indicating management zones and drainage lines (not to scale)

The third wetland system filters overflow from the bottom dam (Plate 4.2). Water runs as uniform surface flow from the bottom dam over about a five metre wide channel, before entering the third wetland. The wetland is a tributary to the Klein Mooi River.

Plate 4.2: View from the bottom dam of the wetland system of tributary joining Klein Mooi River

A set of water tests were taken in the catchment on the study site (Table 4.20). The sampling was conducted according to the sampling guide: *Quality of Domestic Water Supplies* (2000), and the samples were submitted to the Umgeni Water Board for analysis. These tests are not required for the three steps in the proposed guidelines, but were used to get a better indication of the concentrations of solute nitrogen throughout the system. These tests were not meant for conclusive findings, but rather to establish an overall picture of the nitrate, nitrite, ammonia and total kjeldahl nitrogen movement at various points along the catchment.

The levels of nitrate, nitrite and ammonia on the Sourveld Farm at the time of sampling generally did not exceed recommended standards. Dallas and Day (1993) tabulated the current water quality standards of five countries in their review on riverine ecosystems. According to this table, South African nitrate special effluent standards should not exceed 1.5 mg of NO_3-N per litre. Ammonia standards for the protection of aquatic life should not exceed levels of 0.016 mg per litre, although the special effluent standard requires that ammonia levels do not exceed 10 mg per litre (but is currently being revised). Canada bases the ammonia standards on the pH, as increasing temperature and pH will increase the toxicity of ammonia (Dallas & Day 1993). The nitrite standards in Canada and the United States require that levels do not exceed 0.06 mg of NO_2 - N per litre.

Although there was generally no excessive nitrogen pollution to the system evident in the water samples, there was evidence of aquatic plant growth in both the large storage dams (Plate 4.3). The small drainage catchment on the farm and the fertilizer practices still make the farm a good case study for this mini-dissertation.

Plate 4.3: Aquatic plant and weed growth in the Sourveld dams

Table 4.20: Nitrogen concentrations on the Sourveld Farm: 30 October 2001

WATER TEST RESULTS				
SITE	**NITRATE** (mg N/l)	**NITRITE** (mg N/l)	**AMMONIA** (mg N/l)	**TKN** (mg N/l)
A: Top dam entry	0.22	< 0.05	0.11	3.75
B: Top dam exit	< 0.05	< 0.05	0.3	5.17
C: Duck dam entry	< 0.05	< 0.05	6	8.25
D: Duck dam exit	2.01	< 0.05	0.57	5.05
E: Canal exit	0.18	< 0.05	0.09	9.76
F: Klein Mooi entry	0.21	< 0.05	0.04	0.83
G: Bottom dam exit	0.32	< 0.05	0.19	2.18
H: Klein Mooi exit	0.21	< 0.05	0.05	1.8

The sites were chosen to establish entry and exit concentrations through various drainage areas, and are marked in green from A to H (Figure 4.8). Water tests taken before (F) and after (H) the tributary entry into the Klein Mooi River showed no change in the nitrate concentration at the time of sampling. This suggests that the drainage catchment through the farm does not supply significant amounts of nitrate to the Klein Mooi River, but that the third wetland system below the farm attenuates the nitrate before river entry.

The results of the tests taken at the tributary entry to the top dam before study site (A) and the tributary exit at the bottom dam after the study site (G), indicated that nitrate concentrations exiting the farm was 31% greater than nitrate concentrations entering the farm. This small increase in nitrate concentration suggests that the agricultural practices on the farm may not supply excessive nitrate to the system, although as mentioned above some of the nitrate may be attenuated in the wetland system before entering the Klein Mooi River.

An area of concern according to the water samples was a micro-catchment labelled management zone 3 (Mz3). The results of the water tests in this micro-catchment indicated an increase in nitrate concentration from 0.05 mg per litre at the duck dam entry(C), to 2.01 mg per litre at the duck dam exit (D) into the bottom dam. The nitrate concentration of this drainage line at the entry into the bottom dam was ten times greater than that entering the entire catchment at the top dam at the time of sampling. This micro-catchment is similarly identified as potentially high in non-point source nitrate supply in the guidelines (***Step 1***) below.

<u>STEP 1:</u> <u>PRIORITISING MANAGEMENT ZONES</u>

Step 1a). Delineating the catchments

An orthophotograph (1:10 000) of the Sourveld Farm was used to delineate the catchments. During a site visit, the micro-catchment boundaries and drainage lines were confirmed. Of the 16 micro-catchments identified on the farm, 10 micro-catchments contributed water to the river system under study (Figure 4.8). Six micro-catchments supply the top dam with water, three of which predominantly drain the Sourveld Farm. The top dam is bordered by both pasture (perennial rye) and potato plantations. The bottom dam also receives water from six micro-catchments, draining both pasture (perennial rye) and maize fields. The dams share two micro-catchments. The micro-catchments were marked on the orthophotograph, and each became a management zone. These were labelled Mz1 to Mz16.

Step 1b) Determining the potential nitrate loading

Detailed descriptions of each of the 10 management zones (Figure 4.8, labelled 1 to 10) relevant to the mini-dissertation on the Sourveld Farm were made. Information like landuse practices, immediate topography, existing riparian vegetation, and problems associated with nitrate were noted. The various landuse practices for each micro-catchment were identified, and the potential loading for each described. Fertiliser applications for each of the three main agricultural activities on the farm were acquired. The fertilizers contain a mixture of phosphorous, potassium chloride and nitrogen. The nitrogen content is predominantly made up of urea. The perennial rye fields each receive 445 kilograms of nitrogen per hectare per annum, the potato fields each receive 144 kilograms per hectare per annum, and the maize grain fields receive a total of 141 kilograms of nitrogen per hectare per annum. In a study in the United States, it was estimated that only 18% of the total nitrogen fertiliser application will be utilised by the crop product, the remainder will leach into the water resource (Isermann 1991 as cited in Carpenter *et al.* 1998). It can therefore be assumed that a large proportion of the nitrogen will eventually find its way into the river system, even if estimates in the USA exceed South African nitrogen yields..

The management zone of highest priority according to the nitrate loading potential has been labelled management zone 3 (Mz3) (Figure 4.9). This zone was afforded highest priority as it had the highest source density, intensity and magnitude (according to ***Step 1b***), and therefore potentially the highest nitrate loading rate per unit area. Mz3 drains the dairy operations; the work centre; storage facilities; workshop; the homestead and fields of perennial rye grass. Almost all of the intensive farming operations occur within Mz3. Runoff from the dairy and pasture has a high nitrate loading potential, and the animal enclosures and intensive animal operations in Mz3 all contribute toward this. Water tests taken shortly after the first rains indicated that nitrate loading in Mz3 increased by 97.5% from the top of the management zone drainage line to the bottom dam. The increase was most noted after water passed through a small holding dam (duck dam). It is expected that nitrate slowly accumulates in this dam over time. The ammonia levels decreased by 90.5% over the same distance. Ammonia is converted to nitrate in the process of nitrification, which may explain the findings.

Step 1c). Determining the delivery pathways of water

The underlying local geology is mainly the Tarkastad subgroup of the Beauford Group. This subgroup commonly has sandstone and subordinate shales. The wetland areas which are now dammed are mainly alluvium. Water is forced to follow shallower flowpaths by the underlying geology, and it is unlikely that nitrate will bypass the riparian areas at great depths. Mz3 is still considered of highest priority, as its delivery pathway of water and soluble nitrate can be managed by riparian buffer zone establishment.

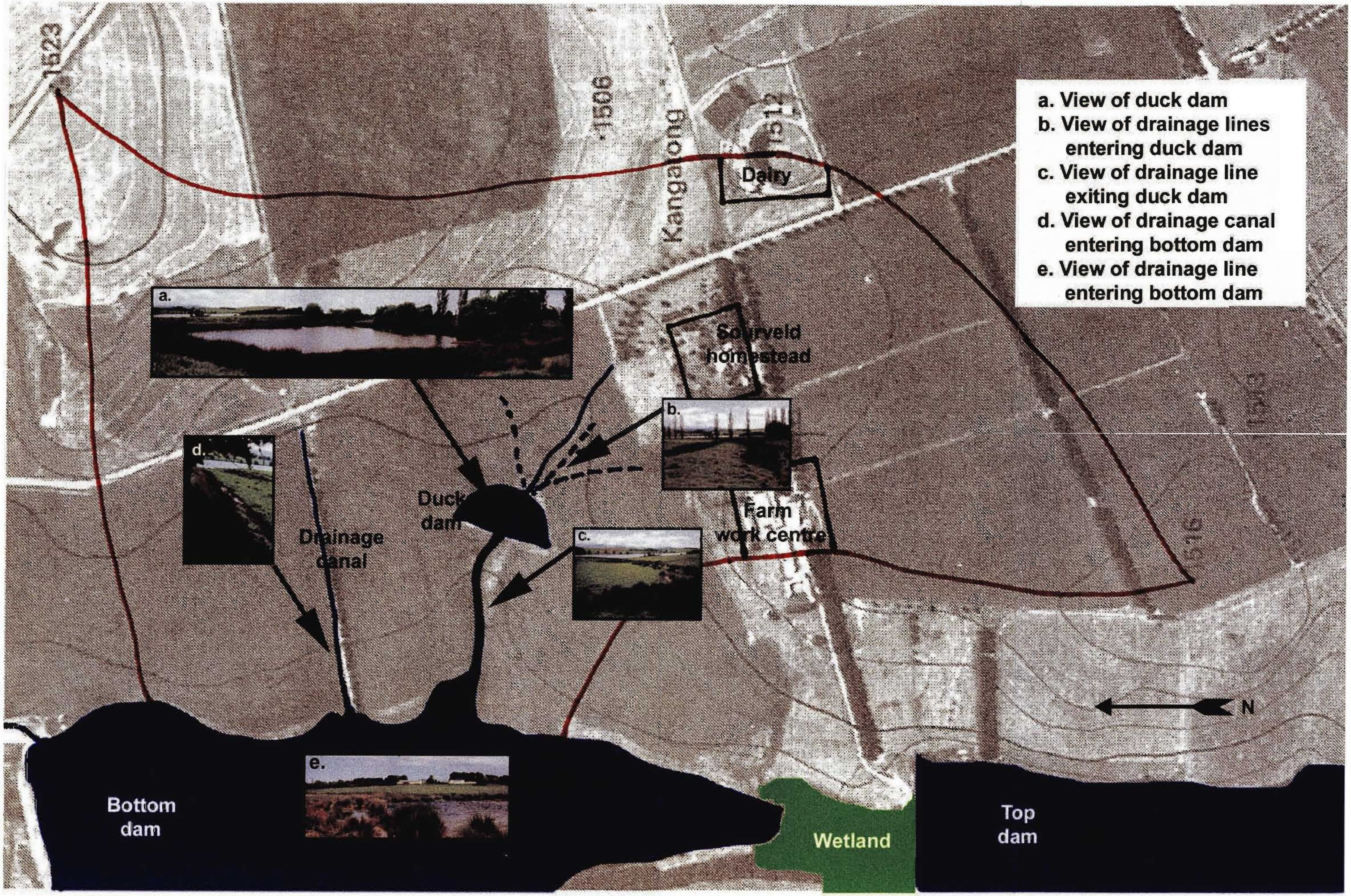


Figure 4.9: Management zone 3 on the Sourveld Farm (not to scale)

STEP 2: ASSESSING THE NITRATE ATTENUATION POTENTIAL

Mz3 was afforded top priority according to ***Step 1***, and will therefore be used to demonstrate the practicality of the following step.

Step 2a). Determining the potential for denitrification

The majority of Mz3 is planted with perennial rye grass, which is grazed by the livestock. The fields are irrigated, and fertilizer is applied approximately once every month. The average annual rainfall in the Kamberg-Kangatong area is > 800mm. The saturation index of the soils is therefore quite high, where the soils are usually saturated. The predominant organic matter content of the soils in Mz3 is very low, as the grass provides little litter for decay, and the soils are frequently tilled.

2a): DENITRIFICATION POTENTIAL (DP)	**= (SI x 0.5)**	**+**	**(OMC x 0.5)**
	= 4(0.5)	+	1(0.5)
	= **2.5**		

Step 2b) Determining the potential for vegetative uptake

Factor 1: Soil permeability

The soils of Mz3 are mainly histosols, and have a deep internal free water occurrence, with a high saturated hydraulic conductivity. The hydrologic soil group they belong to is Group B, and the management zone receives a HSG score of 4. The average slope of Mz3 is 1:40, or 2.5%, and receives a score of 5, as a slight slope will have a very slow velocity of runoff, allowing for good infiltration.

Soil permeability	**= (RO x 0.5)**	**+**	**(S x 0.5)**
	= 4 (0.5)	+	5 (0.5)
	= **4.5**		

Factor 2: Delivery pathway rating

Subsurface water follows shallow flowpaths as the gentle gradient and shallower histosol soils promote higher water tables, and shallow lateral flow. SSF receives a score of 4. Surface run-off in Mz3 from the farming operations is channelled along constructed drainage ditches and under the bordering farm roads. The water enters into the fields and predominantly follows three small natural drainage lines before entering a smaller holding dam (duck dam). The overflow is then channelled along a single drainage line through the perennial rye to the bottom dam. Another man-made linear drainage canal buffered on the one side by a single line of exotic trees enters the bottom dam below the more natural drainage system of the duck dam. This canal drains a well-used farm road which is the main livestock route to the dairy, and is bordered by fields of perennial rye grass. None of the drainage lines in Mz3 are buffered by any form of riparian vegetation, although there are parts of a very small wetland system evident where the drainage systems meet the bottom dam. As the majority of the surface runoff follows drainage lines and constructed drainage ditches, the delivery pathways of surface flow in Mz3 are rated as canals, and it receives a score of 1.

Delivery pathway rating	**= (SSF x 0.5)**	**+**	**(SF x 0.5)**
	= 4 (0.5)	+	1 (0.5)
	= **2.5**		

Factor 3: Vegetation structure rating

The majority of the drainage lines in Mz3 are buffered only by perennial rye grass. There are a few trees scattered mainly as windbreaks along some of the channels and around the small duck dam. The grass has a low rooting depth, high rooting density, very high stem density of groundcover potential, and very low leaf litter potential.

Vegetation structure rating	**=**	**$\sum$(CRITERIA 1-4)**
	=	2 + 4 + 5 + 1
	=	12 (Score in Table 4.11)
	=	**3**

2b): VEGETATION UPTAKE POTENTIAL (VUP)	**=**	**(SP x 0.33*) +(WDP x 0.33*) + (VS x 0.33*)**
	=	(4.5 x O.33*) + (2.5 x O.33*) + (3 x O.33*)
	=	**3.3**

***Step 2c) Scoring the estimated nitrate attenuation potential* (NAP)**

NITRATE ATTENUATION POTENTIAL (NAP)	**= DP + VUP**
	= 2.5 + 3.3
	= **5.9** (Score in Table 4.14)
	= 46% - 55%

STEP 3. **TO DETERMINE ADEQUATE BUFFER WIDTH**

3a) Organic matter content is very low and receives a score of 80.

3b) The soils belong to hydrologic soil group B, with a score of 20.

3c) The slope is slight, at 2.5% it receives a score of 5.

3d) The subsurface flow is shallow, with a score of 10.

TBW **= (OMR x 0.25) + (HSGR RATING x 0.25) + (SR x 0.25) + (SSFR x 0.25)**

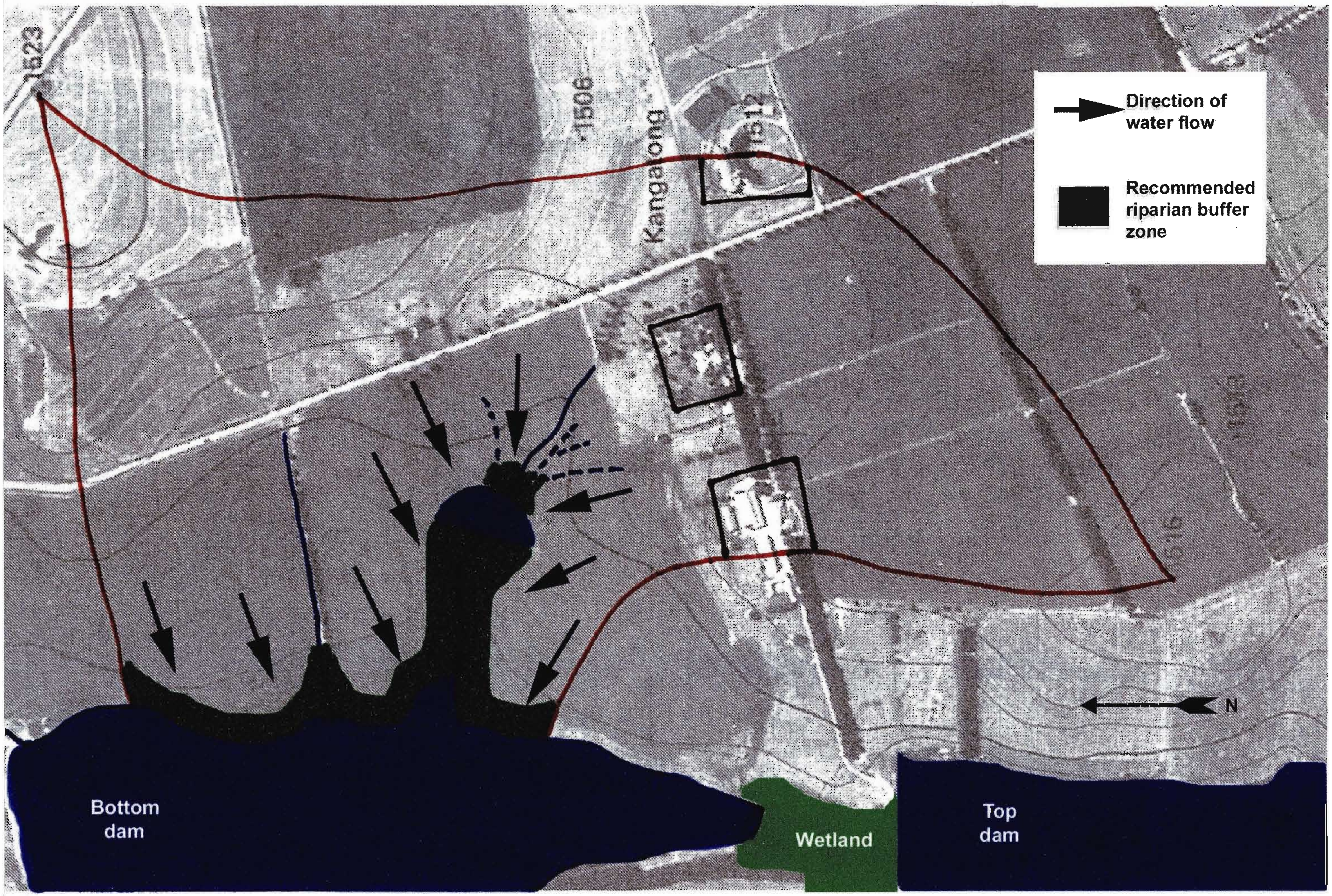
= (80 x 0.25) + (20 x 0.25) + (5 x 0.25) + (10 x 0.25)

= 28.75m

4.5.1 Practical implications of the findings

The case study presented a drainage area that could be divided into 10 management zones. Each of the management zones were prioritised in terms of riparian buffer zone establishment. Mz3 was afforded highest priority according to ***Step 1*** of the guidelines, as the local geology promoted suitable solute nitrate travel for management by RBZ's, and the nitrate loading potential in this management zone was highest. ***Step 2*** of the guidelines indicated that Mz3 had an average existing nitrate attenuation potential, where it is anticipated that only 46% to 55% of the total nitrate loading is attenuated. A drainage canal has been constructed between the farm road and the bottom dam, which interferes with the natural drainage system of the management zone. Soluble nitrate draining into the drainage canal would bypass the riparian area all together. The drainage canal is an earth ditch, bordered on the one side by exotic trees. This would require a slightly different management approach, where ***Step 3*** of the guidelines would probably be ineffective. The natural drainage line flowing through the duck dam to the bottom dam was then assessed by the guidelines in ***Step 3***. A linear buffer zone width of at least 30 metres on either side of the drainage line was deemed sufficient to increase the site potential for denitrification to about 70%.

The linear buffer width recommendation is an estimate, and neglects to include catchment shape. The application of the width recommendation would require logical planning, where site features would indicate how to include catchment shape. For example, the natural drainage line of Mz3 is interrupted by both a small holding dam (duck dam), and a man-made drainage canal. To establish a 30 metre buffer zone on either side of the entire mapped drainage line would be unnecessary. The specific site details will give a clue to where the recommendation should be applied (Figure 4.10).



gure 4.10: Recommendations on riparian buffer width and shape for management zone 3 (not to scale)

As the duck dam acts as a nutrient trap half way along the drainage line, it is recommended that a buffer zone between the duck dam and the bottom dam would be most effective. Water from the duck dam would then filter through a riparian buffer zone of about 60 metres in total width (30 metres either side). This linear width recommendation can be reduced as the drainage line nears the bottom dam. A linear buffer of about 30 metres along the bottom dam boundary of Mz3 would trap any nitrate bypassing the drainage line. Water enters the duck dam from numerous small drainage lines and natural ditches. It is possible to establish a small section of riparian vegetation at this entry point, which would further reduce nitrate levels in the duck dam itself.

4.5.2 Future management objectives for the Sourveld Farm

Management objectives for the function of nitrate attenuation are of particular relevance to this mini-dissertation. However, the variety of riparian functions are of benefit to both the landowner and other catchment users. For this reason, recommendations regarding the riparian function of nitrate attenuation will be complimented with recommendations regarding other riparian functions where possible.

The common management actions applied to agricultural landscapes such as periodic burning to remove dead vegetative material, or the harvesting of material for hay production, or the application of fertilisers and pesticides should not interfere with the individual management of the riparian buffer zone. This zone should be managed individually.

i) Choice of vegetation

Generally, all riparian zones established on the study site should utilise fast growing indigenous plant species, which should be selected to promote both age and species diversity. Palone and Todd (1997) recognise age and species diversity as important in providing adequate leaf litter for microbial processes, adequate groundcover for promotion of infiltration, and potential habitat diversity for many insect and animal species. All of these relate directly to both improved water quality and habitat functions of riparian buffer zones.

Lowrance *et al.* (2000) recommend the three zone buffer system in the United States for the purpose of nutrient attenuation. It was the finding of this mini-dissertation that neither grassed nor forested buffers were particularly more efficient than the other in nitrate attenuation, but that vegetation density and structure was most important. For this reason it is recommended that all riparian zones established on the study site for the attenuation of nitrate should consist of mixed plant species, to allow for a range of stem and root densities, rooting depths, and canopy heights. Indigenous tree species planted adjacent to the drainage lines and dam boundaries will improve bank stability, aquatic habitat, and nitrate attenuation. These trees may be planted along the contour, and growth over time will soon create a more natural forest belt. Smaller shrubs and woody vegetation can be planted adjacent to the trees. This zone will attenuate both soluble and sediment-bound pollutants and promote riparian habitat functions such as biodiversity, as well as improving organic matter content of the soils. These two zones may require periodic harvesting, clearing of dead and diseased trees, and weed or alien invasive control, but generally will require little maintenance.

The three zone buffer system usually recommends an indigenous grassed zone adjacent to the crop product. This zone acts to slow water runoff and trap sediment. Much of the dam boundaries are vegetated with perennial rye grass as a crop product. This grass will fulfill the function of the grassed zone. If indigenous grasses are used in this zone, they may require regular burning to maintain vigorous growth. This burning should not interfere with zone 2, as the burning practice may result in a high mortality of woody vegetation. Areas where other crop products adjoin the dam boundary may require a small filter strip of indigenous grasses before the shrub and tree zones. The grassed zone may require periodic slashing, and non-grazed areas directly adjacent to the shrub zone should be encouraged.

ii) Restricted access

Cohen *et al.* (1987) recommend that livestock have restricted access to streams in agriculturally productive catchments. The Sourveld dams are already fenced off from the pasture and crop products, and livestock watering facilities are provided away from the dam boundaries. Livestock should be excluded from the riparian zones along the dams and drainage lines to prevent trampling, bank instability and overgrazing (Palone & Todd 1997).

iii) Prioritise objectives

The ten micro-catchments contributing to the study drainage area on the Sourveld Farm are all potential future management zones (labelled Mz1 to Mz10 in Figure 4.8). The proposed guidelines (***Steps 1*** to ***3***) may be used to prioritise and assess each of these management zones in terms of nitrate attenuation, as was the example in Mz3 above. However for the purposes of this mini-dissertation, the future management objectives for these zones will be considered broadly, without applying the guidelines to each management zone.

a) High priority objectives

Management zone 3 (Mz3) was afforded the highest priority in the proposed guidelines above. This particular management zone (Mz3) would be the most important zone to concentrate on in terms of riparian buffer zone establishment. A suggestion of buffer width and shape for Mz3 was made in ***Step 3*** above (Figure 4.10).

b) Medium priority objectives

Those management zones with distinct drainage lines should be afforded the next level of priority, as buffer zones may be required along the drainage lines themselves, and along a small linear section of the dam boundaries. Management zone 7 (Mz7) has a very distinct drainage line, which is fenced off from the surrounding fields. At the time of the site visit, the surrounding fields of planted trees had been burnt after tree felling. The remaining stumps and woody debris promotes rilling and more concentrated water channelling. There was water flowing in the drainage ditch, following deep rills and gullies. The drainage ditch was vegetated with natural grasses and reeds. A borrow pit and access road parallel to the dam boundary interrupts the natural drainage of the management zone. It is suggested that the drainage line be buffered with indigenous vegetation, and a larger buffer zone be established at the boundary of the top dam. This buffer zone may be an extension of the adjoining wetland, using quick-growing indigenous wetland species such as reeds, sedges and rushes.

Management zone 8 (Mz8) has a natural depression between two hill slopes, although no actual drainage line is evident. The depression is currently unvegetated, and is used as an access road between the potato crop (Plate 4.4). The water from this area drains down through the depression, where it is intercepted by a bottom farm road running parallel to the top dam boundary. The water is then channelled along this bottom road until it is able to filter through a section of grassland, where it enters the top dam as sheet flow. The depression does not appear to carry large quantities of water, but in times of heavy showers water may flow at high velocities through this section. The depression may require more formal soil erosion control and storm water management techniques to dissipate the energy of the water, reduce sediment travel and promote infiltration. The drainage depression is already a wide channel, and the water is able to disperse. However the soil in the depression is bare, and susceptible to erosion. It is advised that the depression be vegetated with indigenous grasses.

The use of the depression as an access road should be dissuaded, but if this is not possible, the road should be more clearly defined, and runoff made to leave the road along the contour by using engineering techniques such as mitre drains. The road should be a grassed track. A small vegetated buffer at the base of the dip, before water enters the bottom road parallel to the dam boundary is advised. The existing grassed section below the bottom road already acts to slow runoff from the road, and this area is of less priority. A small linear buffer along the banks of the top dam would still be of benefit, however.

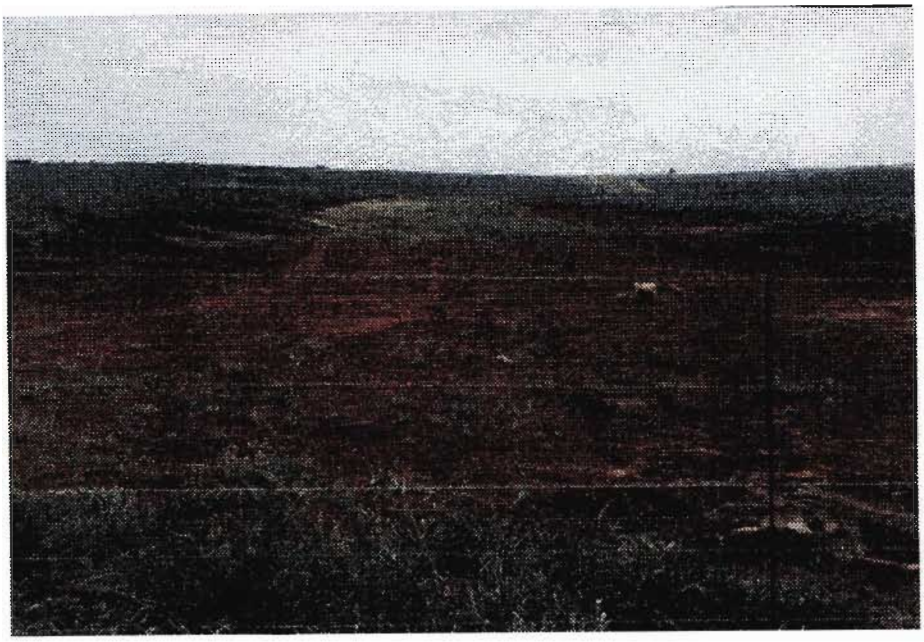


Plate 4.4: View of drainage dip in Mz8, currently used as an access road

c) Low priority objectives

Low priority management objectives are suggested for management zones with an existing high potential for nitrate attenuation. Management zone 1 (Mz1) and management zone 6 (Mz6) drain areas outside of the Sourveld Farm boundaries. The main drainage lines in both of these management zones are buffered by natural wetland systems before entering the top dam. These management zones will therefore not be considered in terms of future management. The wetland systems throughout the farm can be considered within the management objectives. These wetlands are performing an important water quality amelioration function, and the water tests suggest that this function is highly efficient. The water exiting the farm at the Klein Mooi River contains 0.1mg N/l less than that of the water entering the farm, suggesting that the wetlands may be removing nitrate from the system. The wetlands are already fenced off from the paddocks, and are not cultivated, and as such are functioning well. Priority should be given to maintaining the functions of these wetlands, and perhaps managing these wetlands for improved nitrate attenuation. The first two of the three wetland systems on the farm both receive water from culverts. It is suggested by Cohen *et al.* (1987) that culverts should not alter the gradient of stream flow, or the width of the stream. As these culverts are already in place, neither of these recommendations can be met without structural change to the system. Both sets of wetlands receive water from the culverts from suspended outlets (approximately one metre above the wetlands), which are approximately 60cm in diametre. It is therefore recommended that the culverts be lined with stone and concrete (stone pitching) to dissipate the energy of the water flow. Pebbles and boulders placed below the outlets may help to prevent knick-point erosion and scour, and disperse the water flow into the wetland systems.

Management zone 2 (Mz2) is planted with pasture rye grass, and contour ploughing and irrigation is practised (Plate 4.5). The management zone has three natural dips in the hillside, where water may tend to accumulate. Only one of these dips shows evidence of a more permanent drainage line. This drainage line has steeper slopes, and much richer colour to the grass suggesting greater moisture content. However at the time of study there was no water flowing in the drainage line. The drainage line enters the top dam through about two metres of reed and natural grasses. A road and fence line run through the centre of Mz2, to meet the waters edge. Hay production is evident on some fields. The entire top dam is fenced off from the paddocks and fields above. The practise of contour ploughing reduces flow velocity and promotes infiltration throughout Mz2. Most of the water draining from this catchment enters the dam as uniform sheet flow, however some rilling is evident.

Plate 4.5: View of Mz2 from the opposite bank of the top dam, showing two of the three dips in the hillside

Similarly management zone 4 (Mz4) has no identifiable drainage lines, and generally water enters both the top and bottom dams as uniform surface or subsurface flow. A long term goal might be to create a small linear buffer of indigenous vegetation along the entire boundaries of both the top and bottom dams. The second wetland system between the two dams also receives water from this management zone. The wetland system should not be removed and management recommendations made above apply.

Any new developments proposed adjacent to the dams and drainage lines should preserve an undisturbed corridor of riparian buffer zone of suitable width. Vehicle access through the riparian zones should be kept to a minimum. New drainage ditches and canals should attempt to disperse water into the riparian zones before stream entry, as nitrate and other pollutants will bypass the riparian zones in canals that enter the dams directly. Slopes greater than 40% should be included in the riparian zones (Cohen *et al.* 1987). Long term goals should include linking riparian zones to create corridors of riparian habitat. Future management objectives may include the promotion of other riparian water quality and habitat functions in conjunction with nitrate management.

5. DISCUSSION

5.1 APPROACH TO PROBLEM

The variety of water quality problems associated with agriculturally derived non-point source nitrogen and the increasing national and global concern with conservation of water resources, demands that research into water quality conservation be undertaken. This mini-dissertation focussed on the solute nitrate component of nitrogen pollution as an example of how riparian buffer vegetation can be a successful water quality amelioration management practice. The mini-dissertation attempts to illustrate an approach to thinking about management and design of riparian buffer zones for improving water quality, rather than providing a definitive answer to the problem.

The topic of nitrate attenuation in riparian buffer zones has been afforded the attention of many researchers, as riparian buffers tend to be the most influential means of solute nitrate control. The large literature base presented many very efficient riparian buffer zones at nitrate attenuation world-wide. However many studies also found certain riparian environments to be relatively inefficient at nitrate attenuation. It was questionable as to what was promoting mechanisms of nitrate attenuation in certain buffer zones, and what was hindering the function in others.

5.2 UNDERSTANDING THE PROBLEM

There was general consensus on the two key mechanisms of nitrate attenuation in riparian buffer zones, although there was debate as to which of the two mechanisms was more important, and under which circumstances (Section 3.2). However, the literature review in this mini-dissertation suggests that both mechanisms are necessary if nitrate attenuation is to be maximised in riparian buffer zones. It was then necessary to understand what was responsible for providing the optimum riparian environment for these two mechanisms to function at a maximum.

Studies regarding the topic focussed on understanding the problem at various scales. Most studies focussed on those factors influencing the mechanisms at an organism level, some chose to look at the field level factors, whilst others where most interested in the influential factors at a landscape and regional level. For the purpose of this mini-dissertation, it was possible to group the various scale approaches into two, those at an organism level (Section 3.3), and those at a landscape level (Section 3.4). It was anticipated that a database system would provide the best means of comparing the literature, to isolate the characteristics of the riparian environments most responsible for enhancing the factors.

Collating information from the various studies on the topic presented many problems, as many of the studies were so dissimilar. The variety of definitions of riparian environments meant that the actual riparian environments used in the studies varied greatly. The studies were also conducted all over the world, meaning that the different topographies; seasons; climates; soils; vegetation; measurement techniques and units often interfered with gaining the necessary information regarding nitrate attenuation from the studies. The objectives of the studies often differed too, where a variety of methods were used. Thus the lack of common criteria used by these studies make comparisons difficult.

The information from the database was graphed in various ways to try to better see the relationships between the factors and the commonalities in the study sites of high nitrate removal efficiencies. However, as many studies do not make use of the same study criteria, much of the information from the studies was too varied to compare effectively. These comparisons may be more conclusive if the graphs were to be further understood through statistical analysis. It may even be possible to create more graphs of relevance if more information on the study sites is obtained directly from the authors. In this way the relationship between the mechanisms of nitrate attenuation and the factors influencing these mechanisms may be more definitive.

Nevertheless, the graphs created for this mini-dissertation were still able to reflect some important commonalities of riparian nitrate attenuation efficiencies. Three similar characteristics of the efficient riparian environments were noted. Studies where there was shallow lateral subsurface or uniform surface delivery pathways (Section 3.4.1), high density vegetation structure of good composition (Section 3.4.2), high organic matter levels in the soils and fluctuating soil surface saturation rates (Section 3.4.3) showed the most significant nitrate attenuation efficiencies.

It has been suggested that certain vegetation types are more efficient at specific riparian water quality functions than others (Peterjohn & Correll 1984; Osborne & Kovacic 1993; Klapproth 1999). The graphs created in this mini-dissertation indicated, however, that both forested and grassed riparian zones could be equally efficient at nitrate retention, and in some instance neither proved particularly effective (Figure 3.14). This may be a very relevant point to be noted when designing riparian buffer zones. It was more evident from the literature that mixed vegetation proved most efficient at an array of riparian functions. Certainly mixed vegetation provides a variety of root and stem densities, rooting depths, leaf litter compositions and habitats. All of these factors were noted in the mini-dissertation for improving nitrate attenuation in riparian environments. Further to this, a high primary plant productivity was noted as important in providing organic carbon for the process of denitrification, and in utilising nitrate in plant uptake more rapidly. As such, the debate regarding most effective vegetation type for nutrient attenuation may benefit from these findings on nitrate utilisation in riparian zones.

5.3 TOWARDS MANAGEMENT RECOMMENDATIONS

The diversity in existing recommendations regarding riparian buffer zone design for nutrient attenuation, and more specifically nitrate attenuation, makes implementing these zones a difficult process. Inadequate buffer sizing and shape can be costly on the environment, while over-sizing can be costly on the landowner. As previously mentioned, a limitation in developing recommendations is the inconsistency in existing study information necessary for riparian zone comparisons. Although many studies exist regarding nutrient and even specifically nitrate attenuation in riparian buffer zones, they do not all follow the same research procedure or methods, and seldom document the same types of information (for example documenting delivery flow paths, surface gradients and soil saturation levels in all nitrate studies).

This mini-dissertation reflected on existing recommendations, and developed a set of criteria specific to nitrate attenuation. The list of criteria was used to develop guidelines, which are a proposed means of assisting farmers and similar landowners in sizing and shaping riparian buffer zones for management of nitrate pollution. The set of proposed guidelines worked through three steps, and are an example of how information can be used to develop recommendations (Section 4.2). The first step in the guidelines focussed on reducing the area of concern into more manageable units. The drainage lines were used to identify micro-catchments, which would group the similar routes of water movement and nitrate stream entry. The micro-catchments would each become a management zone. The problem with defining the management zones like this, is that the drainage catchments have to be reduced to the smallest level or singular route of water movement. The larger the micro-catchment, the greater the potential for there to be more than one drainage pathway of water, and the greater the chance of nitrate bypassing the riparian buffer zone.

The management zones were prioritised according to the potential nitrate loading and the delivery pathway of water and solute nitrate within each zone. These guidelines did not provide a definitive means of establishing exact nitrate loading potential, but did attempt to assess the nitrate source in each catchment.

The assessment of each management zone of priority in ***Step 2*** of the guidelines (Section 4.3) rated factors established as important through the database exercise. But there are other factors that may need to be included which were not evident in the findings of the mini-dissertation, but may become relevant through further research. The last step in the guidelines was an attempt to show how certain factors could be used to make decisions in riparian management such as the determination of buffer width. The diversity in width recommendations noted in this study indicates the variety of possible factors that could be used for width determination. An attempt was made in this mini-dissertation to sort these factors into a smaller, more definite set. Four factors were chosen through the findings of the mini-dissertation.

These factors were scored and rated. Although backed by scientific support, the ratings have been chosen as relative, normative factors. The rating of these factors was applied to the site characteristics of existing individual studies, and a width estimate was established. This estimate was compared to the actual riparian buffer widths used in the individual studies. Unfortunately, the factors required for ***Step 3*** were not always documented in the existing studies, so only a few studies proved suitable for comparison. The Pearsons Correlation Coefficient was equal to 0,829 and significant at the 0.01 level. This step (Section 4.4) in the guidelines may have neglected other important factors, but when this step was used in existing studies showing good removal efficiencies, it worked fairly well. As such, it was decided that the four factors were sufficient.

If the proposed guidelines were to become an accepted means of assessing and sizing riparian buffer zones specifically for nitrate attenuation, they would have to be refined. The proposed guidelines made use of terms such as 'good', 'moderate', and 'high'. These can be given numerical values to make the rating of the different characteristics easier. The four factors used in the rating system of ***Step 3*** were all covered in some way in the three factors in ***Step 2*** of the guidelines. But ***Step 3*** required that only these four factors be rated and scored. It would be more beneficial to a user if the final score obtained in ***Step 2*** of the guidelines could be used in ***Step 3*** to determine buffer width, rather than re-rating and scoring the factors again. This would however require further research.

The way in which the guidelines can be used was demonstrated in the practical application of the steps in a case study (Section 4.5). One set of water tests were taken to give a better indication of the nitrogen movement through the farm. True testing would require a series of tests taken over a period of time, where loading rates and rainfall patterns are taken into account. However for the purposes of this mini-dissertation, the single set of water tests did provide a sufficient indication of the movement of nitrate; ammonia; nitrite and total kjeldahl nitrogen in the study area. The three steps to the guidelines were applied to the study site. What was evident from this exercise was that the guidelines help to recommend an average linear width of a buffer for a particular site. More complex drainage systems in management zones would require that recommendations be made regarding the shape of the riparian buffer according to the catchment shape. A linear buffer width is not always suited to the drainage system. The guidelines would need to include for both catchment shape and stream order. Management zones along the banks of a fifth order river would require a different shaped buffer to management zones surrounding a first order stream. But if the management zones are sufficiently reduced to single water flow pathways, the linear width recommendation along each management zone might still be effective.

6. CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

The contribution agricultural nitrogen pollution has made to degrading South African water resources has been acknowledged for some time. Many approaches aim to reduce non-point source nitrogen inputs into the system, through reduced fertilisation practices and conservation farming practices. Reduced inputs will improve the nitrogen loading to a system, but still does not manage to reduce the excess nitrogen needed for crop production from reaching the water resources. A complimentary conservation practice of riparian buffer zone establishment will trap and utilise non-point source nitrogen in both sediment and as a solute before stream entry.

The variety of recommendations regarding buffer sizing for this function has made riparian buffer zone establishment a difficult task. For example, Section 2.6.5 showed a buffer width range of five metres to ninety metres recommended as necessary for nitrate attenuation (Castelle *et al.* 1994; Palone & Todd 1997; Dosskey *et al.* 2000). What is evident is that buffer width required for solute nitrogen is larger than that required for sediment trapping. It is acknowledged that management of these zones for improved sediment removal will differ from management of these zones for improved solute attenuation. However the objectives of this research were not to define management techniques, but to simply assess and size buffer zones in an aim to facilitate future management recommendations. As such, buffer zones sized for solute nitrogen control would still be of sufficient width for sediment-bound nitrogen control. The mini-dissertation then focussed on solute nitrogen, with an example in nitrate.

The first objective of this mini-dissertation was to identify the mechanisms responsible for attenuating nitrate within riparian buffer systems. There was a general consensus in the literature that nitrate is utilised by both plants and soil micro-organisms in riparian buffer zones. These two key mechanisms of nitrate attenuation (in riparian buffer zones) have been well researched at an organism level since the late 1970's. Only recently, however, has any time been afforded to those factors influencing these mechanisms at a landscape level. The second objective of this mini-dissertation was to isolate commonalities in riparian environments that may be responsible for enhancing the two key mechanisms of nitrate attenuation.

A data base of the available literature was established for means of riparian buffer zone comparison. There appeared many commonalities at a landscape level in riparian characteristics as a result of these comparisons. Comparisons of studies finding very high nitrate attenuation efficiencies suggested that good riparian nitrate attenuation world-wide is a result of three factors in riparian environments, namely: the delivery pathways of water and solute nitrate, the vegetation structure, and the soils. As the right combination of these three factors at a landscape level would result in very good nitrate attenuation by riparian buffer zones (greater than 65%), it followed that the riparian areas could be assessed according for their nitrate attenuation potential by these factors at a landscape level.

The third objective was to develop a proposed means of assessing and sizing riparian buffer zones to achieve a reasonable removal efficiency (>65%). Guidelines were developed as an example of assessment of existing nitrate attenuation potentials of any given site. The findings of such assessments would be useful for riparian design, sizing, establishment and management for improved nitrate attenuation. This mini-dissertation was unable to explore all the design options for improved nitrate attenuation, or recommend specific guidelines regarding buffer establishment. However the mini-dissertation did look toward answering one of the most common questions in riparian buffer design: how to determine an adequate width for desired function, by suggesting guidelines for the purpose of nitrate retention.

The wide range in the width recommendations (Section 2.6.5; Section 4.4) for nitrogen attenuation in the consulted literature suggested that buffer sizing for nitrate attenuation was a daunting task. The proposed guidelines illustrate an approach to determining buffer width, which incorporates current understanding regarding the factors contributing to nitrate removal. The purpose of this research has been to present these guidelines as an approach rather than an explicit system of assessment, as considerable further testing is required. The selection of factors and their weighting may thus vary as more experimental evidence comes to light. Guidelines such as these can be integrated into a more holistic management strategy for water quality protection, where water users such as Catchment Management Agencies may use information like this in the development of a Catchment Management Strategy.

The final objective was to indicate the practicality of the guidelines in the field. A case study was used to demonstrate the use of these guidelines. As proving the effectiveness of these guidelines through the case study was not possible given the time constraints, the width determination technique was tried out on existing studies. The studies proving the most efficient at nitrate attenuation (over 65% removal efficiencies) were used. Site information from each study was used to follow the steps of the guidelines, and a suitable width for good removal efficiency was recommended. These recommendations were very close to the actual widths used in the studies, indicating the value in the guideline recommendations.

6.2 RECOMMENDATIONS

This research is exploratory in nature and consequently there are several areas for further investigation. Further research may include consideration of similar water pollutants and the use of riparian buffer zones for their control, or perhaps achieving definitive answers about the design; establishment and management of these zones for nitrate attenuation and similar nutrient and sediment trapping. Nitrate pollution is one of many water quality problems, and guidelines such as those established in this mini-dissertation may also be established in a similar way for other pollutants too. The findings of this mini-dissertation may be integrated with similar studies on other riparian functions, both habitat and water quality, in order to develop recommendations suitable in the South African context about the holistic use of these zones.

The recommendations arising from this mini-dissertation are bulleted below.

- The lack of common site information documented by researchers makes any form of riparian studies comparison very difficult. It is recommended that a list of key site criteria be developed, which researchers are encouraged to document in all riparian studies regarding nitrate attenuation.
- A standardised methodology for measuring nitrate attenuation in riparian buffer zones should be created, for means of easier research comparison.
- It is recommended that the database used in this mini-dissertation be extended to include all world-wide literature regarding nitrate attenuation by riparian buffer zones. It is further recommended that this database include other studies regarding water quality amelioration in agricultural landscapes, such as those focussing on sediment and phosphorous.
- Further research into the similarities of effective riparian buffer zones will indicate more conclusive characteristics responsible for enhancing the mechanisms of nitrate attenuation. Similar research may be conducted on the functions of sediment and phosphorous trapping by these zones.
- The proposed guidelines for the assessment and sizing of riparian buffer zones outlined in Chapter 4 require further testing and research. It is possible to compare these guidelines to other techniques used to assess and size riparian buffer zones. Comparison of the guidelines will give a better indication of their usefulness.
- The rating system in the proposed guidelines may be refined, where the user can more accurately rate vegetation structure and composition, and characteristics of the soils such as the depth at which organic matter content should be rated.
- The guidelines can also be applied to a wetland system for means of comparison. The

potential of the wetlands for nitrate attenuation can be compared to the results of water tests showing the nitrate balance for the system.

- The guidelines are aimed at facilitating researchers in approaching topics regarding riparian pollutant attenuation. These proposed guidelines are an example of how researchers can achieve recommendations regarding the sequestering of nutrients. Such recommendations can be made to farmers and similar landowners.
- An instruction manual with a set of guidelines specific to the South African context should be developed. The set of guidelines should assist in the sizing, shaping, establishment, management and monitoring of riparian buffer zones for the specific function of nitrate attenuation.
- Similar research into developing guidelines for the management of other water pollutants derived from agricultural landscapes is also achievable. It is recommended that the design of riparian buffer zones for the attenuation of sediment and phosphorous be considered for similar instruction manuals.

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