REPORT TO THE WATER RESEARCH COMMISSION

AGRICULTURAL LAND-USE IMPACTS ON WETLAND FUNCTIONAL VALUES

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PREFACE

This document is one of a series arising from a project designed to improve the management of wetlands in KwaZulu/Natal. The project includes the following documents:

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KOTZE D C, BREEN C M, and KLUG J R, 1994. A management plan for Mgeni vlei. WRC Report No 501/7/94, Water Research Commission, Pretoria.

KOTZE D C, 1994. A management plan for Blood River vlei. WRC Report No 501/8/94, Water Research Commission, Pretoria.

KOTZE D C, 1994. A management plan for Boschoffsvlei. WRC Report No 501/9/94, Water Research Commission, Pretoria.

OELLERMANN R G, DARROCH M A G, KLUG J R, and KOTZE D C, 1994. Wetland preservation valuation, and management practices applied to wetlands: South African case studies. WRC Report No 501/10/94, Water Research Commission, Pretoria.

EXECUTIVE SUMMARY

INTRODUCTION (1[1])

The unprecedented global decline in the extent of wetlands is worrying because functioning wetlands have many values which benefit society. This review discusses wetland functional values and how they are affected by different land-use practices. Those commonly cited are:

1. hydrological values (water purification; streamflow regulation, including flood attenuation and baseflow augmentation; and groundwater discharge and recharge);

2. erosion control value; and

3. ecological value (maintenance of biotic diversity through the provision of habitat for wetland-dependent fauna and flora).

THE FUNCTIONAL VALUES OF WETLANDS (2)

Water purification (2.1)

Wetlands may contribute substantially to the improvement of water quality by removing sediment, excess nutrients (most importantly nitrogen and phosphorus) and toxicants (including metals, organic pollutants such as pesticides, bacteria and viruses and biological oxygen demand). Wetlands have several attributes that enhance their water purification potential, including:

1. a high capacity for reducing water flow velocity (see flood attenuation) leading to sediment deposition and increased retention of toxicants and nutrients;

2. the shallow nature of wetland waters, leading to high sediment-water exchange and photodegradation of certain pollutants;

3. a variety of chemical processes (both aerobic and anaerobic) that remove certain pollutants from the water. For example, denitrification, which depends on an aerobic/anaerobic interface, is one of the most important mechanisms accounting for nitrogen removal; while adsorption onto mineral sediment appears to be the most important mechanism accounting for the removal of phosphorus;

4. high rates of mineral uptake by vegetation, due to characteristically high productivities;

5. high soil organic matter levels that favour the retention of pollutants such as heavy metals; and

6. microbes that decompose organic pollutants.

[1] Numbers in brackets refer to the relevant sections in the main body of the document.

Streamflow regulation (2.2)

By delaying the passage of water through the catchment, wetlands have value in that they:

1. attenuate (dampen) floodpeaks; and
2. store water at the wetland site, providing enhanced streamflow during periods of low flow (i.e. baseflow augmentation).

Flood attenuation (2.2.1)
Attributes contributing to the characteristically high ability of wetlands to attenuate floods include (1) the frictional resistance offered by wetland vegetation, and (2) characteristically gentle slopes.

Water storage and enhancement of sustained streamflow (2.2.2)
Although many wetlands have been shown to enhance streamflow during low flow periods (e.g. Schulze, 1979; and Scaggs *et al.*, 1991), this is not always so. This effect depends on characteristics of the specific site (e.g. whether or not winter die-back of vegetation occurs). Water storage and streamflow enhancement are influenced by factors that contribute to flood attenuation, as they are closely associated. However, additional factors such as the nature of the soil and vegetation die-back are more important than in flood attenuation.

Groundwater recharge and discharge (2.3)

Although poorly understood, it appears that more wetlands act as groundwater discharge areas than recharge areas (Larson, 1981). Wetlands perched above the regional groundwater table generally recharge the groundwater, while those in contact with the regional groundwater serve as aquifer discharge or throughflow areas. Wetlands acting as discharge zones should not be considered less important than those acting as recharge zones, as this zone may exert considerable influence over an aquifer (O'Brien, 1988).

Erosion control by wetland vegetation (2.4)

Wetland vegetation generally has a high capacity for controlling erosion by: (1) binding and stabilizing sediment; (2) dissipating wave and current energy; (3) trapping sediment; and (4) recovering rapidly from flood damage (Sather and Smith, 1984).

The ecological value of wetlands (2.5)

Wetlands provide habitat for a diverse assortment of wetland-dependent species, many of which are threatened. For example, of the 108 bird species included in the Red Data Book, 36 are wetland-dependent. Biotic diversity encompasses many levels (e.g. genes, species or communities). To simplify biotic diversity considerations, Preston and Bedford (1988) propose that impact on biotic diversity be assessed by examining the effect on biological integrity (naturalness) and populations of threatened species.

Contribution of wetlands to biogeochemical cycling (2.6)

The contribution of wetlands to biogeochemical cycling, particularly in terms of acting as carbon sinks, has been recognized (de la Cruz, 1982). Hammer (1992) suggests that the restoration and creation of wetlands would be effective in decreasing atmospheric CO_2 levels.

THE IMPACT OF INDIVIDUAL AGRICULTURAL LAND-USES ON WETLAND FUNCTIONAL VALUES (3)

Drainage and the production of crops and planted pastures (3.1)

Conversion of wetland to cropland usually involves complete removal of the native vegetation, hydrological manipulation, tillage, and the application of fertilizers and pesticides (Willrich and Smith, 1970). Pasture production has a similar level of impact but is usually less severe because it generally provides better vegetative cover for the soil than does cropland. If the pastures are perennial then this is likely to reduce the impact further, because commonly grown perennial pasture species tend to have higher wetness tolerances than most crops and annual pastures. Also, perennial plants require the soil to be disturbed and exposed less frequently.

The objective of wetland drainage, which is to decrease the volume and retention time of water in the wetland, is directly opposed to the water storage and purification function of the wetland. If a wetland is acting as a groundwater discharge or recharge area, the effect on streamflow regulation may be considerable (O'Brien, 1988). In addition, wetland drainage may detract from the flood attenuation capacity of a wetland, and replacing the wetland vegetation with actively growing temperate crops or pastures increases water use during the critical dry-season flow period. Wetland drainage also indirectly lowers the wetland's hydrological values through: acidification; a lowering of soil organic matter levels caused by oxidation; increased susceptibility to erosion; the release of toxic elements such as uranium; and subsidence caused by the reduced organic matter and water content of the soil.

Crop or pasture production is clearly detrimental to the maintenance of biotic diversity because it involves disruption of the hydrological regime and the total replacement of the native vegetation. Consequently, the habitat value would be lost for the majority of wetland dependent-species.

Grazing of undeveloped wetlands by domestic stock (3.2)

Although permanent wetlands tend to have a relatively low grazing value, temporary or seasonal wetlands may provide important grazing-lands. Grazing animals affect wetland functional values primarily through defoliation, trampling and deposition of urine and faeces.

Effect of grazing on the ecological value of wetlands (3.2.1)

Domestic stock grazing has been widely shown to have a positive effect on the ecological value of wetlands. For example, grazing may significantly reduce the abundance of reeds in some areas, resulting in an increase in the abundance of aquatic plants and waterfowl that utilize such habitat (Duncan and D'Herbes, 1982). The creation of short muddy areas by grazing stock favours mud probing species and grazing of wet grassland favours breeding lapwings (*Vanellus vanellus*) (Gordon and Duncan, 1988). Furthermore, the positive effect of grazing on plant species richness in salt marshes is well documented (Bakker, 1989; Jensen *et al.*, 1990). Grazing has also been shown to have a positive effect on salt marsh invertebrate species. However, if utilization levels were high relative to plant production levels, diversity may be lowered and ecological integrity lost, particularly in wetlands developed under low grazing pressure (Facelli *et al.*, 1989). Extensive reduction of plant cover would be detrimental to many animal species requiring such cover (e.g. flufftails). By increasing soil exposure and consequent evaporation, it appears that grazing may alter the plant species composition by disadvantaging the more hydric species (Kauffman, 1983b).

Effect of grazing on the hydrological and erosion control values of wetlands (3.2.2)
Most wetland soils have inherently low infiltration capacities (Schulze *et al.*, 1989) and consequently have a low potential for losing infiltration capacity through trampling-induced compaction. Many wetlands do, however, have high erosion potentials (e.g. those with the Rensburg soil form). Such soils generally occur under dry climatic conditions, where the erosional degradation of wetlands has generally been high.

Hydrogeomorphological setting and slope also affect the susceptibility of wetlands to erosion. Wetlands in seepage slope and channel or riparian sites are most susceptible. Soil moisture at the time of use has an important influence because the susceptibility to erosion increases when soils are wet (Wilkins and Garwood, 1985).

Mowing of wetlands (3.3)

Although similar to grazing, mowing differs in that the removal of herbage is less uniform and harvesting is restricted to a much shorter period. Plant species diversity tends to be lower than in grazed areas, but wetland mowing has been widely shown to result in a higher plant species diversity than in unutilized wetland (Green, 1980). Timing of cutting contributes to the effect of mowing. In low producing wetlands, autumn cutting was shown to affect species richness more positively than summer cutting, while in high producing wetlands, the reverse was true (Bakker, 1989). Depending on the extent and timing of mowing, animals requiring vegetation cover could be negatively affected, especially if cutting occurs during the breeding season.

Burning of wetlands (3.4)

Reasons why wetlands are burnt (3.4.1)
Fires, largely caused by lightning, have occurred independently of humans in many wetlands (Loveless, 1959; Schmulzer and Hinkle, 1992). Prescribed burning continues to be used for wildlife management, enhancing stock grazing value, reducing fire risk, and assisting in alien plant control.

Effects of sub-surface fires (3.4.2)
Wetland fires include surface fires, where only the above-ground plant parts are burnt, and sub-surface fires, which consume above- and below-ground parts, as well as soil material. In surface fires, wetland plants usually re-establish rapidly from the undamaged below-ground parts and the soils remain largely unchanged physically (Ellery *et al.*, 1989). In contrast, dramatic changes in vegetation and soil may result from sub-surface fires. By burning away the upper soil layers, sub-surface fires may create open water areas, as appears to be the case in the Okefenoke Swamp, USA (Cypret, 1961), and Wakkerstroom vlei (Kotze, 1992a). In the Okavango Delta, sub-surface fires facilitate the change from declining permanent swamp in abandoned channels, to a seasonally inundated floodplain or mixed terrestrial/aquatic habitat (Ellery *et al.*, 1989). Thus, from an ecological point of view, localized sub-surface fires appear to be generally favourable in that they enhance habitat diversity. However, they may substantially detract from the hydrological and erosion control values of wetlands, particularly when they occur in erosion-prone situations, in that they: (1) destroy organic matter and disrupt soil structure, rendering the soil more susceptible to erosion and decreasing the water storage volume of the soil; (2) release trapped nutrients; and (3) destroy emergent vegetation.

Effects of surface fires on hydrological and erosion control values of wetlands (3.4.3)
By enhancing early spring growth of wetland vegetation, burning increases transpirative loss of water from wetlands for the first few weeks of the growing season. In wetlands with dry-season dormant vegetation, burning is also likely to promote evaporative loss of water by removing non-transpiring standing dead material, which would otherwise protect the soil or water surface from radiation and wind exposure. This effect may last for several months if the wetland is burnt in early winter.

Few studies exist on changes in soil nutrients in wetlands following fire. That of Faulkner and de la Cruz (1982) showed that a brief increase in pH and a more prolonged increase in organic matter and Ca, Mg, K and P occurred.

It is commonly held that fire, by reducing litter input into the soil, decreases the organic matter content of soils. However, this is not necessarily so. In *Phragmites australis* marsh, for example, burning generally stimulates below-ground production, leading to increased root detritus production (Thompson and Shay, 1985), which would offset the reduced incorporation of aboveground litter.

Effects of surface fires on the ecological value of wetlands (3.4.4)
Species populations vary in their recovery rate following fire. The snail *Neritina usnea* was found to be most abundant in the year following a fire, while duck species using *Juncus* marsh for nesting were found to prefer marsh burnt at least three years previously (Hackney and de la Cruz, 1976). Although little fire-related research has been done in KwaZulu/Natal wetlands, it appears that a fire return frequency of 2 years is unlikely to have a major detrimental effect on any of the known wetland-dependent species in the humid to sub-humid areas of this region. However, this may be strongly dependent on the presence of unburnt refuges from which recolonization may occur.

Timing of burning is important, with early winter burning adversely affecting winter breeding animal species and summer burning affecting summer breeding species. Late winter/early spring burning is least likely to impact on breeding animals, as very few species are likely to be breeding at this time.

Fires modifying the plant species composition and structure of wetlands, tending to favour those species characterized by winter die-back. A comparison of burnt and unburnt areas in Nylsvlei (Otter, 1992), Memel vlei (pers. obs., 1993), and Ntabamhlope vlei (Kotze, 1992b) suggest that fire may be used to control alien plants.

Damming of wetlands (3.5)

Many of South Africa's wetlands have been flooded by dams as they often provide ideal dam sites. While dams perform certain wetland functions (e.g. sediment trapping and water storage), they are poor substitutes for others. Notably, the habitat required by specialised wetland-dependent species is frequently lost. Where there is a series of dams along a stream, the cumulative effect in reducing streamflow may be considerable, particularly where extraction occurs (Bruwer and Ashton, 1989). Dams can, however, increase dry season flow if water extraction is low and outflow or seepage through the wall occurs. However, irrespective of whether dams increase or decrease dry season flow, the first wet season flows are often retained in the dam because its water level is low at the end of the dry season. This may have a negative effect on both the river biota and downstream users (Bruwer and Ashton, 1989). The bursting of small dams is an additional disadvantage which may contribute to increased flood damage and sediment release.

It has often been observed that water resources could be conserved by flooding wetlands by damming, since transpiration by wetland plants increases water loss to the atmosphere. However, many workers (e.g. Eisenlohr, 1966; Paijmans 1985; Chapman, 1990) have reported evapotranspirative losses from vegetated wetlands to be similar or less than from open water, particularly when the vegetation is dormant.

CONCLUSION (4)

Much information exists concerning wetland functional values and their tremendous worth to society, but most of this is derived from short-term research projects that examine a single process in one geographic location. Extrapolation of these results may therefore be unreliable. Nevertheless, general principles relating to the nature of wetlands and determinants of wetland structure and function allow qualitative predictions to be made.

The water regime is the primary determinant of wetlands. It follows, then, that when assessing the impact of different land-uses, one of the most important factors to consider is the degree to which the hydrological regime is altered. Important factors concerning the nature of the wetland that should also be considered include:

1. susceptibility to erosion (determined by, *inter alia*: soil erodibility, hydrogeomorphological setting and slope);

2. habitat value for wetland-dependent species; and

3. extent and historical loss of wetlands in the surrounding landscape.

Land-uses vary greatly in the impact they have on wetland functional values. Crop production on drained wetland represents the severest impact. This is followed by annual and then perennial pastures. The grazing of undeveloped wetlands has the least severe impact and frequently enhances the habitat value of wetlands. However, where poor grazing management leads to erosional degradation, the loss of functional values may be considerable. The effect of fire depends strongly on the timing and nature of the fire and although substantial loss of functional values may occur, the effect of burning on wetland functional values is often neutral or positive. Dams fullfill certain wetland functions but are usually poor substitutes for others.

By synthesising information concerning the effect of different land-uses on wetland functional values, this review will assist in developing a system for achieving trade-offs between maximising the benefits derived by different wetland users and minimizing the loss of functional values, which benefit society at large. The need for this to be done will increase with the demand for resources.

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1. INTRODUCTION

The world's wetland area has been declining throughout history, due to development and poor land-use practices (Dugan, 1990). Estimates for the USA show that more than 54% of the wetland area has been lost to development, 87% of this being to agricultural development. There is evidence that a similar trend in wetland losses has occurred in South Africa (Walmsley, 1988). In the Mfolozi catchment, for example, Begg (1988) estimated that 58% of the original wetland area had been lost. As the remaining wetland area has steadily declined, society has begun to appreciate the numerous functional values provided by wetlands, which, until recently, have largely been overlooked.

Wetland functions refer to the many physical, chemical and biological processes that take place in a wetland. Where these functions are of value to society, such as the trapping of nutrients, they are termed functional values. In other words, functional values derive from the manner in which wetlands function and are of indirect use to society. Resource values, on the other hand, are of direct use to society in that they provide tangible resources, ranging from land for crop production to suitable sites for bird-watching (Fig. 1).

Those functional values of wetlands most commonly cited in the literature are:

1. hydrological values, which include:

    a. water purification (removal of suspended sediments, excess plant nutrients, and other pollutants);

    b. streamflow regulation (flood attenuation, water storage and enhancement of sustained streamflow);

    c. groundwater discharge and recharge;

2. erosion control value; and

3. ecological value (maintenance of biotic diversity by providing habitat for wetland-dependent fauna and flora).

The contribution of wetlands to biogeochemical cycling has also recently been recognized by some authors (e.g. Hammer, 1992).

The aim of this review is to discuss these values and focus on how they are affected by different agricultural land-uses. Whereas several reviews concerning the functional values of wetlands have been produced, including those of Reppert *et al.* (1979), Adamus (1983) and Sather and Smith (1984), there do not appear to be any reviews on the effects of different land-uses on wetland functional values, despite the importance of this subject.

A very important aspect of functional values not dealt with in this review is their economic evaluation, for which an extensive body of literature exists (e.g. Leitch and Shabman, 1988; Oellermann, 1992). Expressed in economic terms, functional values may be considerable. For example, in the Norfolk and Suffolk broadland of England, where the natural wetland vegetation that protects the river banks from erosion is destroyed, the river banks have to be artificially reinforced at a cost of approximately US$425 per metre of bank (Turner, 1989).

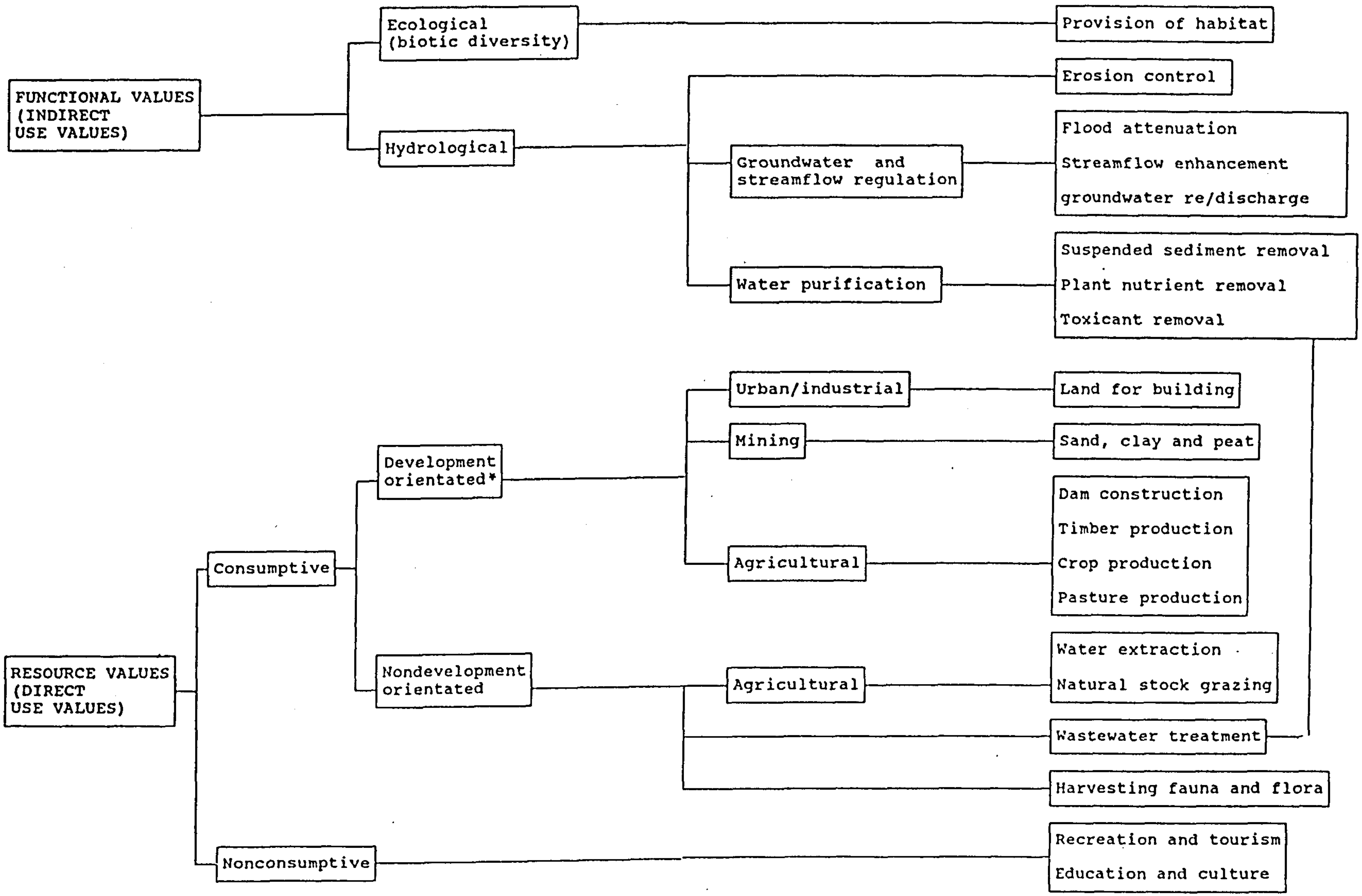

Fig. 1 Use values provided by wetlands.

* Development orientated values require that the wetland be modified by directly removing the indigenous vegetation and/or through hydrological manipulation.

Nondevelopment orientated values do not require that the wetland be modified. However, injudicious management (e.g. heavy grazing leading to accelerated erosion, or excessive extraction of water) may cause extensive modification.

2. THE FUNCTIONAL VALUES OF WETLANDS

2.1 Water purification

2.1.1 Wetland attributes influencing water purification

Wetlands may contribute substantially to improving water quality by modifying or trapping a wide range of substances commonly considered to be pollutants. These include suspended sediment (such as silt and clay), excess nutrients (most importantly nitrogen and phosphorus) and toxicants (e.g. pesticides and excess heavy metals). Excess is taken to refer to concentrations high enough to render the water unsuitable for human consumption. Wetlands have several attributes that enhance their capacity for improving water quality (Kadlec and Kadlec, 1979; Mitsch and Gosselink, 1986; Hammer, 1992) including:

1. a high capacity for reducing the velocity of water flow (because of such factors as the resistance offered by wetland vegetation and the gradual slope of most wetlands) which results in suspended particles being more readily deposited;

2. considerable contact between water and sediments (because of the shallow nature of the water column, leading to high levels of sediment/soil-water exchanges);

3. a variety of anaerobic and aerobic processes, such as denitrification and chemical precipitation, that remove pollutants from the water;

4. the high plant productivity of many wetlands, leading to high rates of mineral uptake by vegetation;

5. high soil organic matter contents (accumulated primarily as a result of anaerobic conditions) which favours the retention of elements such as heavy metals; and

6. microbial decomposition of certain organic substances (such as those introduced through sewage addition). Wetland plants provide substantial surface area for the attachment of microbes, both above-ground and below-ground, due to the aerobic rhizosphere around roots.

Suspended sediments, toxicants and nutrients pass through a wetland as throughflow or are stored for varying periods in wetland storage compartments. In the case of nutrients, these compartments include macrophyte tissue, microbial tissue, detritus, sediments, waters within the soil profile and ponded waters on the soil surface which have a longer residence time than the main throughflow (Howard-Williams, 1983). According to Howard-Williams (1983) the nutrient (or sediment/toxicant) output from a wetland can be calculated as:

Nutrients out = Nutrients in - (transfers into storage compartments - transfers out of storage compartments)

Two points arise from consideration of the above equation:

1. the faster the rate of throughflow (i.e. the more channelled the throughflow) the lower will be the extent of nutrient (and sediment/toxicant) incorporation into storage (Gaudet, 1978; Day et al., 1982, as cited by Howard-Williams, 1983); and

2. although wetland storage compartments have a substantial ability to absorb excess nutrients, they have finite boundaries, and once they are full, there will no longer be transfers into storage. This principle also applies to sediments and toxicants (Howard-Williams, 1983).

A wetland is considered a sink if the input of a given chemical or specific form of that chemical (e.g. organic or inorganic) is greater than the output. Conversely, if output is greater than input, it is considered a source. Through transformation, a wetland may act as a sink for an inorganic form of a nutrient and a source for the organic form of that same nutrient. Determining conclusively whether wetlands are sources or sinks for a given chemical is often hampered by the inadequacy of the techniques used to measure fluxes (Howard-Williams, 1983). In order to calculate nutrient fluxes, water budgets are needed and there are many difficulties inherent in measuring the hydrological components required for water budget determination (Carter, 1986). Thus, even with long term studies it is difficult to assess how efficiently a wetland removes a given pollutant.

2.1.2 Removal of suspended sediment

The higher the mean flow velocity, the greater the ability of water to transport particles of increasing grain size (Hjulstrom, 1935). Flow velocities through wetlands are typically lower than in river channels and the surrounding landscape, and wetlands thus provide important areas where the settling of suspended sediment may occur. Suspended sediment may be detrimental to water quality in itself and it may also carry other adsorbed pollutants (Boto and Patrick, 1979). Turbidity, caused by suspended particles, attenuates light penetration, thereby decreasing photosynthesis (and oxygen production) by submerged aquatic plants. Costly filtration and flocculation processes are generally necessary to free water of particulate matter before it can be used for industrial or domestic purposes (Begg, 1986). High sediment loads are also costly in that they lead to storage capacity loss in dams, an important problem in South Africa (Conley *et al.*, 1987).

Quantitative studies demonstrating the role of wetlands in the removal of suspended solids are lacking. However, one such study in New Zealand (Schouten, 1976 as cited by Begg, 1986) showed that all of the bedload and 50% of the suspended load were being deposited in the wetlands of a particular catchment. Quantitative models exist for evaluating depositional-erosional dynamics, but few studies, except that of Hickok *et al.* (1977), include an identifiable shallow-water component (Adamus *et al.*, 1987).

Qualitative models for sediment trapping are represented in procedures by Reppert *et al.* (1979), Corps of Engineers (1988), Wolverton (1980) and Adamus *et al.* (1987). Included in the procedure of Adamus *et al.* (1987) is a simplified model indicating the gradient necessary to create depositional velocity conditions given different depth and surface roughness categories (Table 1). The most important factor affecting the roughness coefficient is the vegetation - the greater the frictional resistance offered by the vegetation the higher the roughness coefficient. If natural wetland vegetation with a high roughness coefficient (e.g. a dense reed marsh) is replaced by crops which generally have a substantially lower roughness coefficient then this will obviously decrease sediment trapping efficiency.

Table 1 Gradient necessary to create depositional conditions given different depth and surface roughness categories (from Adamus *et al.*, 1987)

Mean Depth (m)	N>0.125[1]	N=0.080[2]	N=0.050[3]	N<0.035[4]
<0.2	<0.0250	<0.0100	<0.0038	<0.0018
0.2-0.3	<0.0150	<0.0060	<0.0023	<0.0012
0.3-0.6	-------	<0.0030	<0.0012	<0.0006
0.6-0.9	-------	<0.0017	<0.0006	<0.0003
0.9-1.2	-------	<0.0013	<0.0005	<0.0002
1.2-1.8	-------	<0.0008	<0.0003	<0.0001
1.8-2.4	-------	<0.0006	<0.0002	<0.0001
2.4-3.0	-------	<0.0004	<0.0002	-------
3.0-3.7	-------	<0.0003	<0.0001	-------

1 Most densely wooded floodplains ("N" is Manning's roughness coefficient).
2 Most densely vegetated emergent wetlands not totally submerged by floodflow.
3 Most moderately vegetated or totally submerged (by floodwater) emergent wetlands, or with boulders.
4 Mostly unobstructed channels.

2.1.3 Plant nutrient removal

In water quality studies, nitrogen and phosphorus are the nutrients most commonly identified as pollutants (Adamus *et al.*, 1987). Wetlands which receive water with high nitrogen and phosphorus concentrations usually demonstrate high removal efficiencies, at least during the growing season (Van der Valk *et al.*, 1979; Begg, 1990). This is considered to be particularly valuable because excess quantities of these nutrients promote algal blooms and population explosions of other undesirable aquatic plants, such as water hyacinth (*Eichhornia crassipes*). These in turn detrimentally affect the suitability of water for domestic consumption and recreational activities (Sather and Smith, 1984).

Freshwater wetlands receive nitrogen and phosphorus from natural sources, such as runoff from vegetated watersheds, and anthropogenic sources, such as effluent discharge, and runoff from fertilized cropland (Hemond and Benoit, 1988). There are three processes by which nutrients are immobilized or removed from wetland waters: (1) accumulation by plants and microorganisms, (2) sedimentation, and (3) denitrification and ammonia volatilization (applicable only to nitrogen). Of these, only denitrification and ammonia volatilization actually eliminate nutrients from the system by releasing nitrogen to the atmosphere. The other two only immobilize and detain nutrients. Nutrients accumulated by plants are temporarily immobilized, after which, they may be re-mobilized or accumulated in the sediment, where they remain immobilized for an indefinite period in an adsorbed or particulate form. Nutrients in the sediment may be re-mobilized and transferred to adjacent waters if, for example, a wetland is disturbed through drainage (Nichols, 1983; Bailey *et al.*, 1985, Howard-Williams, 1985; Richardson, 1985; Richardson and Marshall, 1986).

Denitrification, caused by anaerobic bacteria, is the primary mechanism for nitrogen removal from wetland waters (Sather and Smith, 1984). The denitrification rate varies according to temperature, pH, organic carbon availability, and available surface area. High denitrification rates depend on a continuous supply of NO_3 (associated with aerobic conditions) to anaerobic areas. Wetlands are often suitable sites for this as they are generally characterized by anaerobic sediments (overlain by an aerobic sediment zone, a few millimetres thick), and shallow oxygenated surface water. This, combined with the aerobic rhizosphere that surrounds wetland plant roots, maximizes the aerobic/anaerobic interface where denitrification can occur (Hemond and Benoit, 1988; Hammer, 1992). Denitrification may be enhanced further in wetlands which are alternately wet (anaerobic) and dry (aerobic). High levels of nitrogen loss have been shown to occur under such conditions (Patrick and Wyatt, 1964; McRae *et al.* 1968; Reddy and Patrick, 1984).

Nitrogen may also be removed through uptake by vascular plants and subsequent "burial" when the plants die and organic matter accumulates in the sediments. DeLaune *et al.* (1986) showed that in a freshwater marsh, a large proportion of the nitrogen incorporated in the vegetation accumulates mainly as organic nitrogen in accreted sediment.

Phosphorus immobilization through the development of organic soils is less important than for nitrogen. Richardson (1985) found that wetland mineral soils had a greater phosphorus retention capacity than organic soils. Adsorption of phosphorus onto mineral sediments appears to be the most important mechanism accounting for the removal of this nutrient (Hemond and Benoit, 1988). Phosphorus may also be removed from solution by precipitation as insoluble iron, aluminium or calcium phosphate (Nichols, 1983) or through deposition of suspended sediment to which phosphorus is already adsorbed (Boto and Patrick, 1979). Thus, the ability of a wetland to retain phosphorus through adsorption and precipitation is related strongly to its capacity to trap mineral soils (Hemond and Benoit, 1988) as well as to the particle size distribution of the trapped sediment, which affects the total surface area available for adsorption (Corps of Engineers, 1988). Van der Valk *et al.*(1979) attribute the differences among wetlands in their nutrient-trapping capacity to be primarily the result of differences in hydrology and the interaction of seasonal fluxes of nutrients within a wetland. During the growing season there is generally a high rate of nutrient uptake from the water and sediments by emergent and submerged wetland vegetation. Increased microbial immobilization of nutrients and uptake by algae and epiphytes also leads to retention of inorganic forms of nitrogen and phosphorus. Thus, there is seldom a net export of nutrients during the growing season. Lee *et al.* (1975) consider this pattern to be beneficial because wetlands are most efficient at trapping nutrients during the growing season, the time when the potential for algal blooms to occur is at its highest.

A substantial amount of the nutrients taken up by rooted emergent plants may be lost to the water at the end of the growing season through litter fall and subsequent leaching. However, this is often less than may be expected because, by the time the above-ground parts of higher plants die, most of the nutrients have been translocated to the below-ground storage portions of the plant where they may be "buried" in the deep sediments (Hemond and Benoit, 1988).

Van der Valk *et al.*(1979) list the results of 17 different studies investigating the potential of wetlands to act as nitrogen and phosphorus sinks. These were listed according to whether the wetland in question acted as a nutrient sink for nitrogen and phosphorus and whether this was seasonal. All studies for which phosphorus data are presented indicate that wetlands remove phosphorus from the water passing through them at least during the growing season, and in some cases in all seasons. The same was shown to be true for nitrogen, except for the study conducted by Shih *et al.* (1978 as cited by Van der Valk *et al.*, 1979) which showed that the given wetland acted as a nitrogen source. Overall, Van der Valk *et al.* (1979) conclude that all 17 studies show that wetlands improve water quality to some extent (i.e. in all wetlands there was at least a seasonal net retention of phosphorus and/or nitrogen).

Mitsch and Gosselink (1986) also list the results of 26 different studies of wetlands as nitrogen and phosphorus traps, using the same format as that of Van der Valk *et al.*(1979) and including six of the previously listed studies. The overall results are very similar to those of van der Valk *et al.* (1979) in that in only one of the 26 studies was a wetland shown to be a net source of nitrogen and 4 were shown to act as phosphorus sources.

In summary, Van der Valk *et al.* (1979) conclude that the general picture to emerge from the studies reviewed is that wetlands are always good-to-excellent nutrient traps during the growing season, but in the non-growing season their efficiency declines. Adamus *et al.* (1987) state that few quantitative models exist for evaluating the nutrient retention and removal capabilities of wetlands. Qualitative models include informal guidelines by Kiddy (1979) and more formal procedures by Reppert et al. (1979), Wolverton (1980) and Adamus *et al.*(1987).

2.1.4 Toxicant removal

Toxicants are taken to include metals, organic pollutants, bacteria and viruses and BOD (Biological Oxygen Demand). No specific procedures have been developed for assessing the toxicant removal potential of wetlands, but general principles will be discussed for each group of toxicants.

2.1.4.1 Metals

Metal pollution is often primarily anthropogenic in origin, with the greatest concentrations generally being found in areas with heavy industry or mining (Lazrus *et al.*, 1970). Metal removal efficiencies can vary greatly depending on the particular metals and wetland types involved (Tchobanoglous and Culp, 1980). Giblin (1985) summarized the findings of different studies investigating the passage of metals through various types of wetlands. Measured values ranged from 0% lead passing through an English bog to 100% zinc passing through a North Carolina salt marsh.

Metals may be removed from solution by adsorption onto suspended sediment (mineral and organic), and buried in the sediment when it settles. Metals may also be adsorbed directly onto already immobile sediment (Hemond and Benoit, 1988). The oxidation-reduction (redox) potential is a key factor influencing the retention of metals (Gambrell and Patrick, 1988). Certain metals, such as cadmium and zinc, are more strongly bound to humic material under anaerobic than under aerobic conditions. In contrast, other metals, such as iron (precipitated as ferric oxide under aerobic conditions) may be released back into wetland waters as ferrous iron with the onset of anaerobic conditions (Hemond and Benoit, 1988). The pH is another important factor influencing metal retention.

Most metals are sorbed more efficiently by organic than by mineral soils (Vestergaard, 1979). Since wetland sediments are usually rich in organic matter, they are likely to be better suited for sorption of metals than non-wetland soils with less organic matter. Some metal cations also appear to form organically bound complexes with soil organic matter; in such cases, sorption is essentially nonreversible provided the soil is not disturbed (Wieder and Lang, 1986).

Wetland plants are able to take up metals from the water and sediment. However, the degree to which this leads to the removal of metals depends on the extent to which the plant material is accumulated in organic sediment rather than being exported from the system as detritus (Hemond and Benoit, 1988). Plants may also accelerate the removal of mercury by emission into the atmosphere. Kozuchowski and Johnson (1978) found that there was a positive correlation between mercury emission into the atmosphere by *Phragmites australis* growing on the edge of a mercury-contaminated lake, and concentration of mercury in the sediment.

Another important mechanism by which metals may be removed is through precipitation as oxides, hydroxides, carbonates, phosphates and sulphides. Most transition metals are precipitated as sulphides. This occurs under anaerobic conditions and thus, provided wetlands contain appreciable sulphide ions, the conditions generally prevailing in wetlands tend to promote the precipitation of transition metals. This process is usually more important in saltwater than freshwater because of the generally higher sulphate concentration in saltwater (Hemond and Benoit, 1988).

2.1.4.2 Organic pollutants

Freshwater wetlands may detain and/or chemically degrade organic pollutants, such as pesticides. The two processes may be linked, as when a pollutant is delayed in its passage through a wetland ecosystem long enough to allow degradative processes to occur. One mechanism for the detention of dissolved organic pollutants in wetlands is sorption onto sediments (Hemond and Benoit, 1988). Several different mechanisms may be involved in the degradation of organic pollutants. Wetlands, because of the shallow nature of their surface waters, provide an ideal opportunity for photodegradation to occur (Zafiriou *et al.*, 1984). The degradation of organic pollutants under anaerobic conditions has not been well documented. However, several workers (Parr and Smith, 1976; Sleat and Robinson, 1983; Suflita *et al.*, 1983; Gambrell *et al.*, 1984; Gambrell and Patrick, 1988) have shown that many organic compounds, such as halomethanes, are degraded far more rapidly under anaerobic than aerobic conditions. Thus, wetlands, which characteristically have anaerobic soils, may play a vital role in the degradation of these compounds.

2.1.4.3 Bacteria and viruses

Agricultural and urban runoff entering wetlands may contain large quantities of bacteria, particularly coliforms and pathogens such as *Salmonella* and *Enterococci*, all of which pose a potential hazard to human health. Wetlands have been shown to reduce pathogen counts entering in effluents (Rogers, 1983). DeJong (1976), for example, found bacterial contamination to be greatly reduced by a reed-pond, even during times of peak load.

Several factors may be responsible for the depletion of bacteria and viruses in wetland waters. These include adsorption onto sediments and subsequent sedimentation, exposure to solar radiation, and the presence of toxic substances such as root secretions which have been shown to kill pathogenic bacteria (Seidel, 1970; Rogers, 1983). In addition, one of the most important mechanisms for bacterial removal by wetlands is simply detention while natural die-back occurs. Pathogenic micro-organisms found in sewage effluent generally cannot survive for long periods of time outside the host organisms (Hemond and Benoit, 1988).

2.1.4.4 Biological oxygen demand

BOD (Biological Oxygen Demand) of water is a measure of the oxygen required for the degradation of organic matter. Wetlands decrease the BOD of introduced waters through the decomposition of organic matter during aerobic bacterial respiration (Hemond and Benoit, 1988). While wetland plant material is a source of BOD, the presence of wetland vegetation can also improve purifying capacity by trapping particulate organic matter and providing sites of attachment for decomposing micro-organisms (Hemond and Benoit, 1988).

De Jong (1976, cited by Hemond and Benoit, 1988) studied wastewater purification in a rush pond and found BOD reduction was a function of residence time in the pond. He concluded that removal resulted from infiltration of wastewater into the sediment followed by decomposition by soil bacteria, as well as purification of through-flowing waters by microbes in the pond.

2.2 Streamflow regulation

Wetlands usually have a number of attributes such as gentle slopes, dense vegetation and outflow constrictions that impede the rate of water flow. By delaying the passage of water through the catchment, wetlands have value in that they: (a) attenuate floodpeaks and (b) store water at the wetland site providing a more sustained supply of water during periods of low flow (i.e. they augment baseflow).

2.2.1 Flood attenuation

The ability of wetlands to spread and slow down flood waters, thus attenuating and lagging flood peaks is well known (Chow, 1959; Dugan, 1990). The attributes most often cited as contributing to the effectiveness of flood peak control are:

1. Topography of the wetland site (includes wetland slope and nature of the wetland outlet). Wetlands with constricted outlets or no permanent outlets are considered to have a high potential (Adamus *et al.*, 1987) as are wetlands with a gentle slope;

2. Size. The larger the wetland the greater the area provided for flood storage and velocity reduction;

3. Nature of the vegetation (Plate 1, p13). Tall robust vegetation offers more frictional resistance than short softer vegetation (Table 1). Essentially, the effectiveness with which vegetation attenuates floods is closely related to its effectiveness in sediment trapping, as both are a function of flow velocity reduction;

4. Water regime. The potential for a given wetland to attenuate floodflow is lower if it is already covered with standing water (i.e. if it is flooded) than if it has no standing water; and

5. Permeability of the soil. Soils with a high infiltration potential are considered to have a high potential. However, if the soils are close to saturation then their capacity to take up flood waters is low, irrespective of permeability. Thus, due to the wet nature and inherently low infiltration potential of most wetland soils, this factor is often unimportant in the attenuation of floods.

The U.S. Army Corps of Engineers concluded that a substantial reduction of floodwaters from the 1955 hurricane occurred along the Charles River because of the natural storage effect of wetlands flanking the channel. This contrasts with the far more serious flooding that occurred in the Blackstone River, which is similar but lacks natural storage (Childs, 1970 as cited by O'Brien, 1988).

A quantitative approach to the flood attenuation potential of wetlands was undertaken by Ogawa and Male (1986), who used a hydrological simulation model to investigate the relationship between upstream wetland removal and downstream flooding. The study found that the increase in peak streamflow was significant for all sizes of streams when wetlands were removed. However, although an isolated wetland may perform a significant flood control function, effective control is more often the result of the combined effect of a series of wetlands within a particular catchment (Verry and Boelter, 1978).

2.2.2 Water storage and enhancement of sustained streamflow

A popular belief is that wetlands increase dry season streamflows by acting as sponges which gradually release water from wetland storage (Ingram, 1991). This "sponge model" arose largely out of observed reductions in streamflow perenniality from catchments subject to extensive wetland destruction. Begg (1986) cites the Blaaukrantz River as a good example of this. At its headwaters were numerous wetland areas which gave rise to the river once noted as a clear strongly flowing perennial stream. Over the years the catchment, including the wetlands, became intensively farmed and overgrazed. By 1945 the flow of the river was no longer perennial, nor was the water clear. Unfortunately this example, like others of its kind, suffers from the disadvantage that it is impossible to say to what extent destruction of the wetlands *per se* led to a decrease in the water quality and sustainability of streamflow. This is because the effect of wetland mismanagement is compounded by mismanagement of the catchment as a whole (see Section 3.2.2). What is needed, then, are more rigorous investigations (e.g. comparing the measured outflow from paired catchments, that are monitored).

Schulze (1979) compared the streamflow regimes of two catchments in the Ntabamhlope area, one with very few wetlands and the other with a series of large wetlands. The coefficient of variation of streamflow was lower and the peak flow was two months later in the wetland-rich catchment. Schulze (1979) suggests that the storage effect of the wetlands is the probable reason for the delayed peak flow (Fig. 2).

Scaggs *et al.* (1991) compared continuously measured outflow rates on paired 130 ha sites (an undrained wetland site with native vegetation and an adjacent site that was drained and planted to fescue pasture) on three different soil types. Runoff hydrographs are plotted on Fig. 3 for one of the soil types over a 19 day period that included two significant rainfall events. Scaggs *et al.* (1991) found that for all soil types, peak runoff rates for the developed sites were usually 2 to 4 times greater than those from undeveloped sites. Runoff rates between peaks were substantially lower for the developed sites, clearly demonstrating the regulatory potential of wetlands.

There is, however, conflicting evidence concerning the role of wetlands in enhancing streamflow during low flow periods (i.e. base flow augmentation). Bullock (1988, cited by Ingram, 1991) showed that in Zambia, dry season flow was greater in a catchment with extensive wetlands than in one without. However, in Malawi, Drayton *et al.* (1980) found no significant difference in late dry season flows from catchments with and without wetlands. One of the major explanations for observations that wetlands do not enhance dry season flow is that evapotranspiration in the wetland depletes groundwater reserves, so reducing water available for dry season flow. A dry season water balance was calculated by Bell *et al.* (1987, cited by Ingram, 1991), for a wetland in Zimbabwe indicating that the volume of dry season flow is only 20% of the evapotranspirative losses. However, in wetlands with relatively cold dry seasons, such as those that occur in the Highland Sourveld (Acocks, 1953) almost complete die-back of the vegetation occurs in the dry season. Unless burnt, the standing dead material in these wetlands would greatly retard water loss.

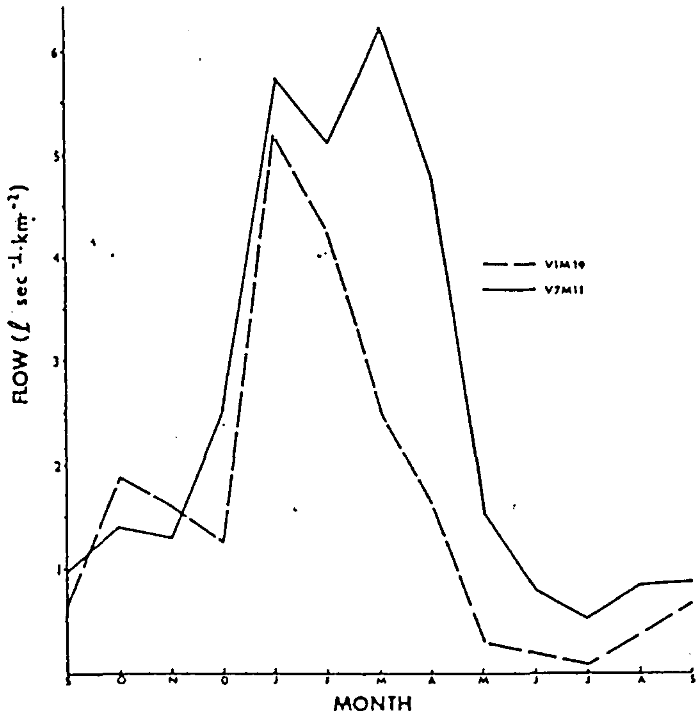

Fig. 2 Comparisons of mean monthly streamflows from a wetland rich catchment (V1M19) and a wetland poor catchment (V7M11) (from Schulze, 1979).

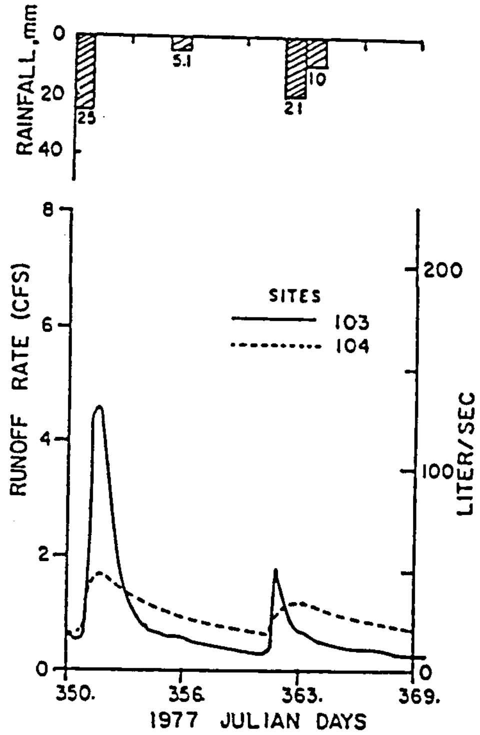

Fig. 3 Runoff hydrographs from a natural (104) and a developed site (103) in a North Carolina wetland (from Scaggs *et al.*, 1991).

The effect that wetlands have on enhancing sustained streamflow during low flow periods is influenced by much the same factors that contribute to flood peak attenuation. However, where the nature of the soil may have little effect on flood attenuation, it is frequently an important factor contributing to the enhancement of sustained flow. Wetlands tend to have a high organic content in the upper soil horizons which increases the porosity and water holding capacity of these layers as well as the overall depth of the soil profile (Begg, 1986; Mitsch and Gosselink, 1986). These factors may be important in contributing to a wetland's ability to withhold water (Angus, 1987). Not all wetlands have a high organic content, particularly those that are infrequently saturated. If this is the case and, in addition, the soils are of a shallow nature, the extent to which these wetlands would enhance streamflow during low flow periods is likely to be negligible.

In conclusion, empirical evidence shows that the popular belief of wetlands as sponges that are able to "squeeze themselves out" during dry periods is untrue. Nevertheless, evidence, such as that produced by Schulze (1979) and Scaggs *et al.* (1991), shows that wetlands do potentially have a regulatory effect by slowing down the runoff process. However, this may be offset by evapotranspirative losses if the wetland vegetation remains actively growing during the dry season. Caution should be observed in drawing conclusions from comparisons of catchments, because observed differences and/or similarities in the timing of runoff are not only a function of the wetland itself but also of the topography, soils and temporal distribution of rainfall within the wetland's catchment. Clearly, our level of understanding is not adequate to predict the regulatory effect of a given wetland with confidence. Over and above those factors already discussed, additional factors, such as the extent to which the wetland is acting as an aquifer discharge or recharge area, need to be considered.

2.3 Groundwater recharge and discharge

The role that wetlands play in groundwater recharge and discharge is poorly understood. While it is agreed that some wetlands act as recharge areas, most occur where water is discharging to the surface (Carter *et al.*, 1978; Larson, 1981). The relationship of wetlands and groundwater is largely a function of their hydrological and topographical position as well as their underlying geology (O'Brien, 1988). Hydrological position refers to the position of the wetland relative to the main zone of saturation (O'Brien, 1988; Winter, 1988).

Generally speaking, wetlands perched above the main zone of saturation (the upper limit of the regional groundwater) are in a position to recharge the groundwater, while those in contact with the main groundwater zone of saturation serve as aquifer throughflow or discharge areas. In addition, some wetlands may change during the course of the year from acting as a recharge area to acting as a discharge area.

Even if a wetland acts as an aquifer discharge area, it may exert as much influence on ground water aquifers as a wetland acting as a recharge zone. Freeze and Witherspoon (1967), for example, have indicated that in small aquifers where the water table is near the surface, the recharge area tends to be large in proportion to the discharge area. Therefore, a wetland that overlies a discharge area is in a position to exert considerable control over groundwater discharge. The effect of a groundwater discharge wetland on hydraulic head distribution can be shown by digital models developed from a US Geological Survey finite difference model (Trescott *et al.*, 1976, as cited by O'Brien, 1988). While the model is subject to certain limitations, it does illustrate the potential importance of a discharge wetland in influencing head distribution and flow pattern within an aquifer.

2.4 Erosion control by wetland vegetation

Wetland vegetation plays three major roles in erosion control: (1) it binds and stabilizes soil, (2) it dissipates wave and current energy and (3) it traps sediment (Carter *et al.*, 1978) (Plate 2). Wetland vegetation has evolved under conditions of frequent flooding, and species such as *Phragmites australis* have a high capacity for binding sediments as well as for recovering rapidly from physical damage caused by flooding. The extent to which wetlands dissipate wave and current energy depends on the hydraulic resistance of the vegetation (Table 1). The efficiency with which wetlands trap sediment is linked to the dissipation of wave and current energy, and depends on the growth-form and distributional pattern of the wetland plants.

Clark and Clark (1979, as cited by Sather and Smith, 1984) state that determining the erosion control value of vegetation in a given wetland is complicated by numerous factors. By way of a general summary they conclude that effectiveness depends on the particular plant species involved (e.g. its flood tolerance and resistance to undermining), the width of the vegetated shoreline band in trapping sediments, the soil composition of the bank or shore, and the elevation of the toe of the bank with respect to mean storm high water.

2.5 Ecological value (maintenance of biotic diversity through the provision of habitat for wetland-dependent species)

As is the case globally, the wetlands of South Africa provide habitat for a wide variety of plant and animal species, many of which are threatened. For example, of the 108 bird species included in the Red Data Book (Brooke, 1984), 36 are wetland-dependent (Goodman, 1987) (Plate 3). Species diversity is just one of many levels in the biological hierarchy at which biotic diversity may be described, including: genes, individual organisms, populations, subspecies, species, communities, ecosystems and landscapes (Noss and Harris, 1986). Biotic diversity is also commonly described at different spatial levels. In the case of species diversity, alpha diversity is the number of species within a habitat, beta diversity the turnover of species between different habitats and gamma diversity the turnover within a habitat from one area to the next (Bond, 1989).

In order to simplify biotic diversity assessment, Preston and Bedford (1988) propose two main management goals: maintaining populations of particular valued species, and maintaining biological integrity (i.e. the naturalness of the region). Species are generally considered valued if they are rare or endangered, but may also be valued for commercial, recreational or aesthetic worth, or if they are recognized for their critical roles in regulating the structure and function of ecological communities (i.e. keystone species). Biological integrity refers to the fauna and flora that are characteristic of a region and their relative abundances in the absence of human intervention (Karr, 1987). Human intervention refers to actions that markedly alter driving forces already affecting ecosystem structure and function (e.g. herbivory, fire and flooding regime) or introduce new driving forces such as landfilling and excavation. Assessing valued species is fairly clearly defined and involves determining the degree to which populations of any threatened species are being positively or negatively affected. However, evaluating the biological integrity of a wetland is far less clearly defined. Weller (1988) and Harris (1988) have discussed factors influencing diversity and suggested a number of indicators.

Several changes occur in the biota in response to stress resulting from human intervention (Preston and Bedford, 1988). Stress-induced changes may include loss of higher trophic levels, leading to shortened foodchains and loss of habitat specialists that create faunal and floral identity for an ecosystem or landscape. These changes result in a truncated biotic assemblage heavy with generalists. Thus, any measure of ecological integrity should be sensitive to changes in both the composition and structure of ecological communities. A multiparameter index for assessing biotic integrity using fish communities has been developed and is now being used successfully in water resource assessment and planning (Karr, 1987). Twelve different parameters are used to summarise the status of a community in terms of species richness, trophic composition, species abundance and condition (i.e. patterns and processes from population, community and ecosystem level are examined). Karr (1987) proposed that a similar procedure be used to develop an appropriate index of ecological integrity for wetland species assemblages for different wetland ecoregions. This will need to account for seasonal, year-to-year, and longer-term cycles characteristic of different wetlands (Karr, 1987) (Plate 4).

Since an excess of water is the dominant factor affecting the plant and animal communities in a wetland (Cowardin *et al.*, 1979), and if resources for directly assessing ecological impacts are very limited, a general assumption can be made that the greater the disruption of the hydrological regime, the higher will be the impact on the ecological values.

In both valued species and biological integrity assessments, it is important that the contribution of wetlands to biotic diversity be considered on a landscape level (Preston and Bedford, 1988). For example, wetlands occupying 7% of a study area in the highlands of KwaZulu/Natal accounted for 22% of the small mammal population (Bowland, 1990). The diet of certain carnivorous mammals, such as the serval (*Felis leptialis serval*), that range widely across the landscape, consist almost entirely of small mammals. Thus, even though serval are not considered to be wetland-dependent species, wetlands provide them with an important food source. Harris (1984) suggests that the primary factors in the landscape mosaic influencing biotic diversity are total habitat area, the size-frequency distribution and quality of habitat patches, and the distribution of these patches in relation to each other and to drainage patterns in the landscape.

Preston and Bedford (1988) propose that a landscape level standard be developed empirically from current and historical data on the size and distributional characteristics of habitats within the area subject to evaluation. Development of the standard would need to take into account the relatively short time span of historical data, and natural fluctuations in wetland size and distribution. The growing body of literature on the consequences of habitat loss and fragmentation could be included to estimate the direction and magnitude of changes in biotic diversity to be expected from the disturbance. The relative functional value of individual wetlands (based on their type, size and location) in maintaining biotic diversity at the landscape level could then be qualitatively estimated (Preston and Bedford, 1988).

It is evident from the literature on South African wetlands that there have been no attempts to measure either between-system or within-system diversity and to understand the mechanisms regulating diversity. As it is not possible at present to develop a strategy for the conservation of biotic diversity based on knowledge and understanding of local systems, an intuitive approach offers the only real prospect for wetland conservation (Breen and Begg, 1989).

Breen and Begg (1989) propose that without a technique for classification there is little hope for the formulation of a comprehensive strategy for the conservation of biotic diversity of the wetlands of South Africa. Therefore, the most urgent need is the development of a classification system that is both comprehensive and efficient at identifying the elements of diversity (Noss, 1987). Breen and Begg (1989) suggest that the Nature Conservancy System, as described by Noss (1987), appears to be an effective means of achieving this.

The major components of the Nature Conservancy System are a "fine-filter" for species inventory (with the aim of maintaining populations of valued species) and a "coarse-filter" for community-type inventory (with the aim of maintaining biological integrity). The system is best understood as a set of filters designed to capture as much of the biological diversity as possible. An ideal goal for a State Heritage Programme, for example, might be to protect the best examples of each major community type in each physiognomic region in the state (Anderson, 1982 as cited by Noss, 1987). By recognizing the major community types, the coarse filter is expected to preserve perhaps 85-90% of the species complement of a state without having to concentrate on each species individually. Species that fall through the coarse filter (generally those that occur in only a few examples of recognized community types) are captured by the fine-filter of threatened and endangered species classification. In KwaZulu/Natal, some wetland community studies have been undertaken (e.g. Downing, 1966) for certain areas, but this would need to be extended over the whole province with a uniform approach being applied.

2.6 The contribution of wetlands to biogeochemical cycling

The effect of wetlands on biogeochemical cycling on a global scale is poorly understood and often overlooked. It was only recently that the value of wetlands as major sinks for carbon was recognized (de la Cruz, 1980). Substantial amounts of carbon are currently stored in wetlands and continue to be incorporated into storage. The oxidation of this carbon, caused by wetland drainage, is certainly of global significance (Armentano, 1980; de la Cruz, 1982; Gorham, 1992), especially in view of rising atmospheric CO_2 levels.

The importance of wetlands as sulphur sinks appears to also be of global significance (Hammer, 1992). Sulphur, which is a major constituent of acid precipitation, is far more readily immobilized in wetlands than in most other habitats. Sulphates entering wetlands are reduced to sulphides which react with metallic ions to form insoluble immobilized substances (Hammer, 1992). Thus, Hammer (1992) suggests that redressing some of the atmospheric imbalances caused mainly by the combustion of fossil fuels would be more effectively achieved by restoring and creating wetlands than by establishing non-wetland forests. He draws attention to the fact that the formation of much of the planet's fossil fuel reserves resulted from the immobilization of carbon in wetlands and subsequent transformations.

3 THE IMPACT OF INDIVIDUAL AGRICULTURAL LAND-USES ON WETLAND FUNCTIONAL VALUES

3.1 Drainage and the production of crops and planted pastures

3.1.1 Effects on the hydrological and erosion control values

Intensive agriculture has dramatic impacts on wetland hydrological values, and usually also detracts from the erosion control value of wetlands. The conversion of wetland to cropland is probably the most severe agricultural impact and usually involves removal of the native vegetation, hydrological alteration (typically but not always limited to drainage) (Plate 5), tillage and the application of fertilizers and pesticides (Willrich and Smith, 1970). While fertilization alone can lead to increased levels of nutrients in receiving waters, Hemond and Benoit (1988) suggest that the hydrological alterations associated with cropping and pasture production have the most profound influence on wetland water quality functions (Fig. 4).

The impacts of crop and pasture production on wetland functional values are fairly similar in as much as they both involve removal of the native vegetation, application of fertilizers and disruption of the hydrological regime. However, the impacts associated with pasture production are likely to be less severe since pastures generally provide better cover to the soils than crops (Table 2). Even if flooding occurred when the crops were fully established and cover was at its maximum, the cover provided would be lower than that offered by pastures or native wetland vegetation. If the pastures are perennial then this is likely to further reduce the impact further because:

1. the perennial pasture species commonly grown on hydric soils in KwaZulu/Natal (notably, tall fescue: *Festuca arundinacea*) tend to have greater tolerance to impeded drainage than most crops and common annual pastures such as ryegrass (*Lolium multiflorum*). As such, they require the water table to be lowered less than would otherwise be necessary for crop production, and thus they do not disrupt the hydrological regime as much (Scotney, 1970); and

2. cropping and annual pastures involve frequent (usually annual) disturbance and exposure of the soil, associated with cultivation, whereas perennial planted pastures require replanting only after several years. This has particular relevance to wetland areas prone to erosion (e.g those with the Rensburg soil form). Scotney (1970) recommends that these areas remain permanently under well managed natural vegetation. He adds that under very exceptional circumstances (including almost level slope gradients, considerable width, irrigation and effective management) the establishment of permanent pastures may be permitted. A further important consequence of frequent cultivation is the increased oxidation of soil organic matter due to the exposure of fresh soil surfaces to the atmosphere. As a result of this, carbon and nitrogen levels are generally much lower under systems of annual pastures or crops than under perennial pastures (Miles and Manson, 1992). Miles and Manson (1992) report data from Cedara, KwaZulu/Natal, where the soil organic carbon content of annual pasture was a third of that in perennial pastures.

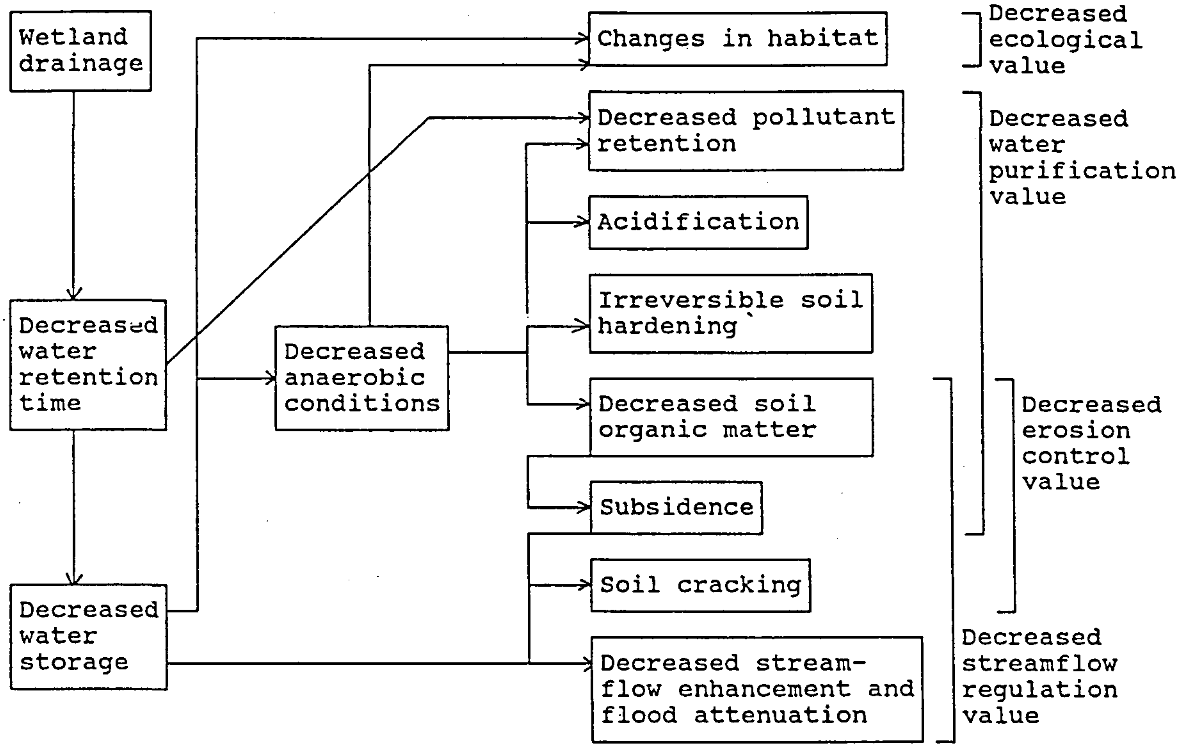

Fig. 4 A conceptual diagram showing the direct and indirect effects of drainage on wetland functional values.

Table 2 The degree to which various activities associated with cropping and annual and perennial pastures tend to detract from the hydrological and erosion control values of wetlands

ACTIVITY	CROPS	ANNUAL PASTURES	PERENNIAL PASTURES
Soil disturbance	- - - -	- - -	-
Reduction in plant aerial cover	- - -	- - -	-
Reduction in degree of wetness	- - - - -	- - - - -	- - - -
	- - -	- - -	- - -

The degree to which hydrological and erosion control values are lost

-	low
- -	moderately low
- - -	moderately high
- - - -	high
- - - - -	very high

3.1.1.1 Direct effects of wetland drainage

For both crop and pasture production, the objective of wetland drainage is to control water flow so as to decrease the volume and retention time of water in the wetland (Scotney, 1970). Most commonly, drains within the wetland are used to decrease water retention times. Additional measures, such as peripheral cut-off drains, the straightening of stream courses and the construction of levees, are sometimes used to reduce the volume of water entering the wetland.

The capacity of a wetland to enhance water quality is directly related to the extent to which flow is directed through the wetland and retained long enough for exchanges to occur with the wetland soil/sediments (Whigham *et al*, 1988). Considering this fact, it is clear that the objectives of wetland drainage are in conflict with optimizing the water storage and water quality enhancement function of a wetland. Wetland drainage, by decreasing the retention time and volume of water in wetlands, also leads to a reduction in its value for storing water and enhancing sustained streamflow. In addition, replacing the natural wetland vegetation (which in the case of the Highland Sourveld is essentially dormant during the winter) with actively growing temperate crops or pastures is almost certain to increase water use during the critical dry season period (Nanni, 1970).

The effect of wetland drainage on flood attenuation is less clear. The flood attenuation capacity of a wetland is dependent on a number of different factors, one of which is the storage capacity in the soil (Section 3.1). If the water table lies at the soil surface at the time of a flood event then the wetland will have no capacity for attenuating the flood peaks by withholding some of the flood waters in the soil. Thus, it is argued that by artificially lowering the water table, the capacity for taking up flood waters in the upper horizons of the soil will be enhanced. However, this argument does not consider that soil usually has a relatively minor contribution towards flood attenuation and that other factors are often more important.

For example, in a wetland associated with a stream, consideration needs to be given to the threshold reached when the capacity of the channel is exceeded and overbank flooding occurs. It is at this point, when flood waters are forced to flow overland, that a wetland is most effective in slowing down the flow rate as a result of such factors as the frictional resistance provided by wetland vegetation. Straightening and/or deepening a river course and creating additional drainage channels result in this threshold being elevated, decreasing the wetland's effectiveness in regulating all those flow events that fall below this threshold. Once flooding of the wetland has occurred, features associated with intensification, such as drainage channels and reduced surface roughness, increase the speed with which water drains from the wetland, and decrease its attenuation capacity. However, if the topographic setting and outlet elevation of the wetland are unaltered, a large proportion of the flood attenuation capacity will be retained, particularly for very large flood events.

Due to the important influence that groundwater discharge and recharge wetlands may exert over the regional groundwater, the effect of wetland drainage on regional groundwater should be considered. The destruction of an aquifer discharge wetland may lower the head potential within an aquifer, which could lead to a decline in the water table and a readjustment of groundwater gradients. This may be critical: it has been shown that a small decline in the groundwater level can lead to a cessation of streamflow (Goode *et al.*, 1977; O'Brien, 1977; Ivanov, 1981, as cited by O'Brien, 1988). The impact of drainage on recharge wetlands may also be considerable and is determined by the extent to which the wetland was previously contributing to groundwater recharge, and by the degree to which the retention time and volume of water in a wetland is decreased by drainage.

3.1.1.2 Indirect effects of drainage caused by a change in the soil environment

In addition to its direct impact, wetland drainage also has unfavourable effects on the soil which, in turn, lower the hydrological value of the wetland. By reducing the duration of soil saturation, drainage causes the soil environment to become more aerobic (i.e. oxidising) which affects the suite of *in situ* processes typically associated with waterlogged soils. For example, sulphides and ferrous iron, formed under anaerobic conditions, may be oxidised to free sulphuric acid and ferric iron respectively, increasing soil acidity (Ingram, 1991). If high levels of iron deposits are present in the soil then increased oxidation and the consequent formation of iron oxides may result in the irreversible hardening of the soil to form a laterite carapace ("ouklip") (Ingram, 1991). This dramatically decreases the effective depth of the soil, which would obviously have considerable hydrological and ecological impacts as well as lowering agricultural potential.

The increased oxidation following wetland drainage results in a decline in the soil organic matter content and this may have a multitude of potentially negative effects. These include reduced water holding capacity, increased susceptibility to erosion, deterioration in soil structure, a decrease in the effectiveness with which heavy metals are trapped, and subsidence of the soil (Lavesque *et al.*, 1982; Ingram, 1991). It has been shown generally that in the first few years following drainage of organic soils, rapid subsidence is caused mainly by drying, settlement and other physical agents. This initial subsidence is followed by a slow but continuous subsidence due to organic matter oxidation (Stephens and Spier, 1970; Lavesque *et al.*, 1982). Mineral soils with high *n* values (i.e. soils with high water contents under field conditions) are also prone to subsidence due to the removal of water and have a low potential for bearing loads (Pons and Zonneveld, 1965; Soil Survey Staff, 1990).

Brinson (1988) contends that drainage and other forms of wetland hydrological manipulation should be seen in the landscape context. Uplands are intrinsically erosional landforms and tend to export most elements (including nutrients, toxicants and sediments). Wetlands, however, are generally importers of elements because they are intrinsically depositional landforms. Thus, wetlands are likely to have a significant impact at the landscape level on elemental constituents in water. When a wetland is drained, eroded or otherwise deprived of its sedimentary function, it exports rather than imports elements. Such alterations normally change the direction of elemental flux from net import to net export.

Brinson (1988) emphasises that wetlands should not be assessed merely for how much nutrient and toxicant retention function is lost, but for how much nutrient and toxicant loading and potentially polluting effect is produced within a catchment unit. Certain wetlands have inherently high polluting potentials. For example, 46% of Colorado wetlands sampled by the U.S. Geological Survey contained moderate (20 ppm) or greater concentrations of uranium (some as high as 3000 ppm) based on dry weight (Owen and Otton, 1992). Disturbance of these wetlands may release the uranium and other loosely bound elements contained in the wetland sediments, particularly if it involved drainage and the resulting oxidation of sediments rich in organic matter (Owen and Otton, 1992). The amount of wetland sediment exposed to more oxidising conditions is one of the most important factors affecting the rate of elemental export from drained wetlands. On exposure, elements that have been accumulating for millennia may be released within several decades. Brinson (1988) cites such occurrences in the Florida Everglades (Stephens, 1956), California (Weir, 1950) and England's East Anglia Fenlands (Hutchinson, 1980). Brinson (1988) recommends that wetlands should be assessed not only for their capacity to trap sediments, which may be slow, but for their vulnerability to export when hydrologically altered, which may be potentially high.

3.1.1.3 Sustainability of wetland crop and pasture production

Despite all the potential negative effects of wetland drainage and intensification, the potential sustainability of crop production on dambos has been fairly widely demonstrated (e.g. Ratray *et al.*, 1953; Elwell and Davey, 1972; Whitlow, 1991) as has pasture production on drained wetlands in the upper Mgeni catchment, KwaZulu/Natal (Scotney, 1970). Dambos are seasonally waterlogged, gently sloping, treeless wetlands containing a natural drainage channel. In Zimbabwe, sustainable cultivation of dambos extends back to before the nineteenth century. Cultivation practices at that time have been documented by Scoones and Cousins (1991). Ridges were constructed parallel to the streamflow direction if the dambo was too wet and required some drainage. If conservation of water was the objective, ridges would be constructed at an angle or along the contour. The central wettest area would be left under dense natural vegetation. Maize would usually be planted on the ridges and rice in the depressions. In relatively high rainfall years the rice would usually be successful and the maize would fail, but in dry years the opposite would generally occur.

These traditional wetland cultivation methods tend to be less disruptive of wetland functioning than intensive commercial cultivation. Traditional crop varieties, for example, are not only quick maturing and pest resistant but some are also more flood tolerant. This contrasts with cultivation of high yielding varieties which require irrigation or a regular water supply and also need heavy fertilizer and pesticide applications (Kolawole, 1991).

Many traditionally used wetlands in Zimbabwe, particularly in the climatically dry areas, are utilized with very little or no drainage (Scoones and Cousins, 1991). The need for wetland drainage is minimized by:

1. cultivating areas which are generally not excessively wet; and

2. multi-cropping with species having different flood and drought tolerances, and scheduling planting correctly.

In addition, impact is also minimized by applying wise soil conservation practices such as green manuring. Thus, rather than attempting to regulate the system completely, traditional wetland cultivation methods tend to account for extremes of the system (i.e. management practices are adjusted to suit the system rather than the system's being altered to suit management requirements).

Rattray *et al.* (1953) cites two examples of contrasting response of wetlands to cultivation in order to illustrate the importance of correct management practices. In a dambo subject to continuous wheat cultivation (resulting in the break-down of organic matter) and ploughing across waterways, degradation of the resource occurred. However, in another dambo where organic matter levels were maintained by green manuring, and where manuring and fertilizer applications and sound soil conservation practices were applied, fertility had been sustained 20 years after cultivation.

In conclusion, two final comments concerning wetland cultivation deserve consideration: (1) even if a given land-use is sustainable (from the point of view of maintaining productivity and having "acceptably low" soil erosion rates), substantial loss of functional values may occur; and (2) when commenting on sustainability, short time horizons are used. Scotney (1970), for example, made conclusions concerning the sustainability of land-use practices that had been in operation for 30 years. However, the system may be slowly declining, and over 30 years this would not be detected without a thorough investigation.

3.1.2 Effects of pasture and crop production on the ecological value of wetlands

Conversion of a wetland to cropland or planted pastures involves disruption of the hydrological regime and the total replacement of the native wetland vegetation. Clearly, this is detrimental to the maintenance of biotic diversity. As is the case with the construction of dams, the altered habitat often attracts species previously not occurring in the wetland. However, these are usually commonly occurring generalist species. For example, *Praomys natalensis*, a species frequently associated with human-induced disturbances, was shown to be absent in an undeveloped wetland but present in an adjacent wetland planted to introduced pasture species (Bowland, 1990).

For the majority of valued wetland-dependent species, such as the long-toed tree frog (*Leptopelis xenodactylus*) and white-winged flufftail (*Sarothrura ayresi*), the habitat value of the wetland would be completely lost following drainage. Although other valued wetland-dependent species, such as wattled cranes (*Grus carunculatus*), continue to use drained and converted wetlands for feeding, drainage renders wetlands unsuitable for breeding (Plate 6).

3.2 Grazing of undeveloped wetlands by domestic stock

Wetlands, particularly temporarily or seasonally wet grasslands, may provide highly productive grazing-lands for wild and domestic grazers (Cooper *et al.*, 1957; Richardson and Arndt, 1989; Findlayson and Moser, 1991). Marsh areas tend to have a lower grazing value because of the relatively unpalatable nature of most mature marsh plants and the excessive wetness and softness of the soil in certain marshes, which prevents access. The high proportion of indigestible structural material is usually the most important factor rendering marsh plants unpalatable. The cell wall component of *Typha domingensis*, for example, has been shown to comprise over 70% of the dry weight of the plant (Howard-Williams and Thomson, 1985). However, young growth of certain marsh species, such as *Phragmites australis*, provide good forage for domestic stock. In its young stages, *P. australis* has a high crude protein-fibre ratio (23%:31%) and no known secondary compounds (Duncan and D'Herbes, 1982).

The two primary components of domestic stock grazing that affect wetland values are: (1) defoliation (and to a lesser extent, uprooting) of plant material as the animals feed; and (2) trampling (through hoof action) of the soil surface and plant material. Other less obvious components of grazing that would also have an effect are: (1) the deposition of urine and faeces; (2) the removal of nutrients and organic matter through meat and milk harvesting; and (3) loss of consumed organic carbon into the atmosphere through animal respiration (Jensen *et al.*, 1990).

3.2.1 Effect of grazing on the ecological value of wetlands

For many wetlands, grazing by wild herbivores has had important effects on ecosystem structure and function (Westhoff, 1971; Bakker, 1978, Gordon and Duncan, 1988). Many wetlands now lack the large indigenous herbivores that once used these areas. It is often not feasible to reintroduce these animals, but domestic stock offer a practical alternative for enhancing the biological integrity of these systems (Gordon and Duncan, 1988). However, grazing may also substantially detract from the ecological value of wetlands, particularly in those developed under low use by indigenous herbivores and in wetlands where utilization is very high relative to plant production.

Grazing by domestic stock has been shown to have a significant effect on the plant species composition and structure of *Phragmites australis* reed marshes in the Camargue (Basset, 1980; Duncan and D'Hebes, 1982) and European salt marshes (Bakker, 1989; Jensen *et al.*, 1990). *Phragmites australis* is sensitive to grazing because the meristem is at the internodes. Both horses and cattle in the Camargue feed readily in water up to 1m deep. When shoots are bitten off below water level, rotting may set in and cause the death of the shoot. Heavy grazing was shown to lower shoot density significantly from 120 to 2 shoots per m^2 and shoot height from 710mm to 160mm (Duncan and D'Hebes, 1982). Basset (1980) also found grazing to diminish *P. australis*. This removal of reeds increases the amount of open water favouring submerged aquatic plants. However, in certain areas, reduction of *P. australis* by grazing does not maintain open water but leads to the dominance of *Typha* and tall *Scirpus* species. These are resistant to grazing, either because their meristem is at or below ground level, or because they apparently contain secondary compounds unpalatable to grazers. (Duncan and D'Hebes, 1982).

Many duck species feed largely on submerged aquatic plants such as *Potamegeton* species. Thus, where grazing activity leads to a decline in reed abundance, allowing for an increase in aquatic plant abundance, these ducks would be favoured. By maintaining short vegetation, cattle also favour waders. Grazing effects on sward height in less hydric wetland areas may also have an important influence on the suitability for certain bird species. For example, black-tailed godwits (*Limosa limosa*) and lapwings (*Vanellus vanellus*) nest in wet grasslands grazed by cattle during the previous summer and avoid that which is ungrazed (Gordon and Duncan, 1988). Sheep grazing produces a finely structured sward ideal for redshank (*Tringa totanus*) (Thomas, 1982).

The creation of mud puddles, reduction in tall dense cover and maintenance of short vegetation areas by grazing stock improves the habitat for mud probing birds such as the Ethiopian snipe (*Gallinago nigripennis*). The largest concentrations of these birds are often found in heavily grazed wetland areas (Neely, 1968) (Plate 7). Prolonged heavy grazing leading to the removal of tall dense cover would, however, disadvantage bird species such as grass owl (*Tyto capensis*) (which require such cover for nesting) and flufftails (which require it for nesting and foraging). If depletion of tall reeds occurs on a large scale throughout a given wetland, it would be detrimental to those species, such as the bittern (*Botaurus stellaris*), requiring reed habitat.

Studies of domestic stock effects on ground nesting birds have shown that trampling can cause the direct destruction of many nests of such birds as lapwings (*Vanellus vanellus*) (Beintema, 1982; Duncan and D'Herbes, 1982). Duncan and D'Herbes (1982) contend that while this occurs at high stocking densities ($>$4 cattle/ha), at stocking densities of less than 3 cattle/ha, damage to nests is likely to be rare, unless the animals are driven in round-ups.

The decrease in plant species richness following exclusion of livestock from salt marsh has been well documented (Bakker, 1990; Jensen *et al.*, 1990). Livestock exclusion may also result in dramatic changes in the invertebrate species composition. For example, halophytic invertebrate species typical of salt marshes may be largely replaced by generalist inland species (Anderson *et al.*, 1989 cited by Jensen *et al.*, 1990). In this case, livestock clearly enhance the habitat value for wetland-dependent invertebrates. However, the ecological benefit derived from grazing "low" salt marsh, situated at the seaward extremity or on saltmarsh islands, is smaller than that derived from grazing the higher parts of the salt marsh. Grazing is not required in "low" salt marsh by species favouring short vegetation (e.g. wading birds) because these areas have inherently short vegetation. Soils are less stable than in higher marsh, and destruction of the turf leads to an increase in bare areas and a decrease in plant diversity (Bakker, 1990).

Very little work has been undertaken in KwaZulu/Natal wetlands to determine the effect of stock grazing and trampling. However, comparison of differences between two adjacent sedge marsh areas in Ntabamhlope Vlei that had been subject to different grazing treatments, allows some tentative conclusions to be drawn for that area (Kotze, 1992b). When compared with the ungrazed area, the grazed area was found to have:

1. a less dense and less uniform aerial cover (provided by the dominant species, *Carex acutiformis*);

2. a higher occurrence and greater extent of exposed mud puddles; and

3. greater plant species diversity, probably as a result of the decreased cover which allows for the establishment of creeping semi-aquatic plants such as *Ludwigia palustris*, and disturbance which favours such species as *Echinochloa crus-galli*.

However, observation of these same areas in winter during a severe drought year showed the plant species diversity to be lower in the grazed area (Doyle, 1992). In the ungrazed treatment, which had abundant litter protecting the soil, the upper soil layers were found to be moist, but in the grazed treatment, which had far less litter, they were dry. It is suggested that the difference in species richness between grazed and ungrazed areas may in part be due to the indirect effect of grazing on soil moisture which affects the growth of more ephemeral species (Doyle, 1992). Kauffman *et al.* (1983b) report a decreased abundance of more hydric species and an increased abundance of species more adapted to drier environments in grazed moist meadows. They also suggest that this is due to increased soil moisture resulting from greater litter accumulation in ungrazed moist meadows.

In addition, Kauffman *et al.* (1983b) found that herbage removal altered the seasonal phenology of moist meadow plant communities, by hastening the onset of anthesis in most species. They suggest that the dense litter layers accumulated in the ungrazed areas probably kept soil temperatures below levels of initiation of growth for longer periods of time.

Trampling by domestic stock often causes wet organic soils to become more tussocky, which may increase sediment microhabitats (Jensen *et al.*, 1990). It is also claimed by Downing (1966) that in sedge meadows in KwaZulu/Natal, cattle trampling causes a very pronounced tussock/channel microtopography with high tussocks (usually > 30 cm high and 50 cm in diameter) (Plate 8). Downing hypothesises that cattle trample the wet clay soil into depressed paths which form a close, criss-crossed pattern. Vegetation in the paths is killed and in time, as cattle continue to use the same paths, they deepen to form channels. The large tussocks (hummocks) are the untrampled areas between the channels. The channels act as drains and because the tussocks are higher and have a larger surface area exposed to the air, they become drier. Martin (1960) also ascribes this tussock-channel formation to trampling by livestock.

Downing (1966) provides only speculative evidence to substantiate this claim. While cattle may be partly responsible for deepening the channels, other factors, such as building by ants and earthworms and the inherently tussocky growth form of some of the commonly occurring plant species appear to be more important (West, 1949; Kotze, 1992b). Observations of hummocks in Mgeni Vlei and Ntabamhlope Vlei, showed ants to be present in some hummocks and earthworms to be very abundant in many of the hummocks. Also, some *Cyperus unioloides* plants growing in hummocks had vertically orientated rhizomes with new growing points positioned several centimetres higher than older points (Kotze, 1992b). This sequence suggests that these mounds have been increasing in height.

An important factor determining the response of a wetland to grazing is whether the wetland developed under low or high grazing pressure. The seasonally flooded Pampa wetlands in Argentina, for example, developed under low grazing pressure. Grazing of these wetlands by domestic stock results in the dominance of cool season species, mainly exotic dicotyledonous plants of low growthform, and the replacement of large tussocks by small tussocks (Facelli *et al.*, 1989). It is suggested that the increased drought risk in summer, caused by trampling-induced infiltration reduction, disadvantages warm season plants (Facelli *et al*, 1989). Grazing caused an increase in diversity at a small scale (5 m) but decreased it at a larger scale. Cool season species were found to be uniformly dominant in grazed areas, but in ungrazed areas, warm season species dominated some patches, and cool season species, other patches. Facelli *et al.* (1989) concluded that in the absence of grazing, different competitive equilibria may occur in the different patches, probably due to subtle environmental differences. In the grazed area, the effects of domestic stock may override the environmental heterogeneity and prevent the achievement of competitive equilibria.

In contrast to the above example, domestic stock grazing may enhance micro-habitat heterogeneity, provided that the grazing intensity is intermediate (i.e. animal utilization levels are not high relative to plant production levels) (Plate 9). Bakker (1990) reports that in grazed salt marshes, the sward structure tends to have less standing dead material and a higher leaf:stem ratio leading to greater digestibility of the forage. This attracts the animals back to the previously grazed areas, even if the overall area has a fairly uniform potential palatability. Thus, if the plant production in a given area exceeds the utilization (consumption and trampling) then a pattern of closely-grazed and lightly grazed (roughgrass) patches usually develops. This grazing pattern is likely in most wetland types subject to such levels of utilization. However, if the level of utilization is high relative to forage production, this usually results in closely grazed swards with hardly any differentiation in the structure of the vegetation (Dijkema, 1984; Bakker, 1989). It can be appreciated, then, that the ratio of closely grazed and roughgrass area could be altered to suit management objectives by changing the level of utilization.

The effect of grazing on wetland communities is not only dependent on stocking rate and timing but also on the type of grazing animal. Van Deursen and Drost (1990) found that grazing of *P. australis* marsh by horses resulted in shorter and thinner shoots and more secondary shoots per primary shoot than that grazed by cattle at a comparable stocking rate. This suggests that horse grazing has a heavier impact and causes a lower-level equilibrium in reed dominance than cattle grazing (Van Deursen and Drost, 1990).

In summary, the effect of grazing on the ecological value of wetlands depends on many factors, such as the intensity and timing of grazing, type of animal, and whether or not the wetland developed under the influence of natural grazers. Generally, grazing enhances the ecological value by maintaining short vegetation areas, giving rise to a greater variety of habitats. In Europe, several conservation organizations encourage extensive grazing. However, these benefits may be lost if the level of utilization is high relative to plant production, particularly if the wetland developed under low grazing pressure.

3.2.2 Effect of grazing on the hydrological and erosion control values of wetlands

Heavy grazing pressure has been shown to have detrimental effects on the hydrological state of wetlands. In the high altitude areas of Lesotho, for example, these include: disruption of flow patterns by paths, gully erosion, an increase in the number of "dry islands" (these features being a function of a change in the hydrology of the wetland), silting up of pools, and encroachment of marginal vegetation into the wetland areas (Institute of Natural Resources, 1991). In

KwaZulu/Natal, the wetlands most severely affected by heavy grazing are those in sub-humid to semi-arid areas, which tend to be more prone to erosion than those in humid areas. While the impact of heavy grazing pressure is often fairly conspicuous, the effect of light or moderate grazing pressure is likely to be far less dramatic.

3.2.2.1 Effect of grazing animals on soil infiltration

Gifford and Hawkins (1978) reviewed the available literature for information useful in understanding the hydrological impacts of grazing intensity as related primarily to infiltration and runoff. The conclusions were that it is difficult to differentiate between the influences of moderate and light grazing. On more porous soils, moderate/light grazing reduces the infiltration rates to approximately 75% of the ungrazed condition, while heavy grazing reduces it to about 50% of the ungrazed condition. This reduction is caused primarily by soil compaction resulting from trampling. Reduced infiltration, in turn, results in higher surface runoff and more rapid loss of water from the catchment. With increased runoff, streamflow response is more rapid, flooding increases and recharge of groundwater storage falls with the result that baseflow yields also fall (Ingram, 1991). Increased runoff also increases the risk of soil loss through surface wash and rill erosion. Soil compaction may also substantially reduce plant growth (Jensen *et al.*, 1990) which further increases susceptibility to soil erosion. Three important characteristics of soil susceptible to compaction are: a low clay content, a high fine-sand fraction and a low organic matter content (Burger *et al.* 1979).

It is important to note, however, that most wetland soils in KwaZulu/Natal have inherently high runoff potentials and, hence, low potentials for losing infiltration capacity. In a list of hydrological information by soil form and series (McVicar *et al.*, 1977) for South Africa, Schulze *et al.* (1989) list the runoff potentials of all soil series. Of the four runoff potential classes, all wetland soil series are given as falling into the highest runoff potential class. Temporarily and seasonally saturated wetland mineral soils tend to have inherently high runoff potentials and low susceptibilities to compaction because of their characteristically high bulk densities and high clay contents, particularly if the clays are expansible. Organic soils and permanently saturated mineral soils, on the other hand, tend to have a low bulk density and a high field capacity. However, under such prolonged saturation conditions, the capacity of these soils for absorbing more water is limited, resulting in their having high run-off potentials. The high percentage volume of water in these soils when saturated allows soil particles to flow as a viscous liquid when trampled, avoiding compaction (Hillel, 1980).

This situation in wetlands contrasts with many non-wetland soils that have inherently high infiltration potentials, which may be lost through mismanagement. It appears then that soil compaction leading to decreased infiltration and groundwater input is more commonly a feature of injudicious grazing practices in the surrounding wetland catchments than in the wetland areas themselves. This emphasises that reduced perenniality of streamflow is often more a function of catchment mismanagement leading to reduced infiltration, than of wetland mismanagement *per se*. Thus, maintaining a sustained water supply requires more than simply managing wetland areas correctly.

3.2.2.2 Effect of grazing animals on soil erosion and soil structure

Although compaction of most wetland soils appears not to be of major concern in KwaZulu/Natal, accelerated soil loss within wetland areas is a major threat to the continued functioning of certain wetlands. Wetlands with steep slopes and soils having a high erosion hazard are the most vulnerable to excessive soil erosion. In KwaZulu/Natal, high grazing pressure leading to severe gully erosion has caused the loss of a large proportion of these wetlands. The most erodible soils generally occur under relatively dry conditions (i.e. mean annual rainfall <800 mm p.a.) (Plate 10). Under more humid conditions (e.g. in the Highland Sourveld of South Africa) soils are generally less erodible. As such, loss of wetlands through gully erosion has occurred considerably less in these areas. In addition, the decrease in basal cover associated with heavy veld utilization is often substantially greater in low rainfall areas and this makes these areas more prone to erosion. Furthermore, rainfall erosivity also tends to be higher in many of the low rainfall areas of KwaZulu/Natal.

Soil moisture content at the time of use may have an important influence on soil loss due to erosion. Generally speaking, when soils are wet they become more susceptible to compaction (Bayfield, 1973; Bryan 1977). They are also more susceptible to hoof penetration, resulting from repeated trampling, which leads to soil truncation and the disruption of soil structure. Such soil is said to be poached or puddled and is rendered more vulnerable to erosion (Bryan, 1977; Wilkins and Garwood, 1985; Vallentine, 1990). Consequently, the likelihood of excessive erosion occurring from seasonal or temporary wetlands would be reduced by confining grazing to periods when the soils are not wet. Hoof action may also destroy leaves, growing points and roots and deposit mud on the herbage, rendering it less palatable. The remoulding and dilation of soil which occurs in poached soils, allows more water to be held in the surface layer. Not only does this reduce its load bearing strength, but it also increases the time taken for soil strength to recover (Wilkins and Garwood, 1985). According to Wilkins and Garwood, the susceptibility of an area to soil poaching is dependent on:

1. Soil texture. Fine textured soils are more at risk than course textured ones;

2. The vegetation type. Certain plant species (particularly those that provide good ground cover and have a high resilience to trampling) afford greater protection to the soil than other plants;

3. The age of the sward. This applies to planted pastures, with recently established pastures being more susceptible than older pastures;

4. Stocking rate. The relationship between stocking rate and severity of poaching is clear, with severity increasing with stocking rate;

5. The grazing system. If a multi-camp rotational system includes wetland camp/s with high susceptibility and non-wetland camps with low susceptibility then a rotational system would obviously provide the flexibility permitting the exclusion of grazing from the wetland camps at the appropriate times. However, where non-wetland camps are absent, there is not consensus in the literature as to whether short intense periods of utilization with long rests are preferable to longer periods of less intense use with shorter rests; and

6. Type of grazing animal. Evidence suggests that sheep cause less damage by deep trampling than do cattle because they have a lower static load (the ratio of animal biomass to total hoof area) than cattle (0.7-0.9 kg cm^{-2} compared with 1.3-2.8 kg cm^{-2}). Impact is greater when animals are moving because in addition to vertical compression, there is horizontal rotary force when the hoof leaves the ground, and there are shear and kick components. It follows, then, that management directed to moving animals slowly and peacefully would reduce the impact of trampling.

Because of their self mulching properties, vertic soils (e.g. those in the Rensburg form) are very crumbly when dry. On steeply sloped areas, at the side of a gully for example, these soils are unstable, particularly if there is traffic over them. Large amounts of soil crumble away from the steep face when dry and would be washed away later by stormflow. Sustainable utilization of these soil types requires that stock be excluded completely from the steep disturbed areas and that the undisturbed areas not be grazed when wet or when sufficiently dry to cause cracking (i.e. these soils have a narrow soil moisture range suitable for use) (Swindale and Miranda, 1981).

Soils with a very high organic content (e.g. soils of the Champagne form) are also considered to have a high erosion susceptibility. Where organic matter-rich soils occur in large wetlands (> ca 50ha) on gentle slopes, the inherent capacity of these wetlands to regulate grazing is high because of:

1. the relatively low palatability of marsh vegetation; and

2. the excessively wet and soft nature of the soils, which limits access by domestic stock.

However, those organic-rich soils occurring in small seepage slope sites are usually characterized by steeper slopes and easier access for domestic stock because the soft soil layers are shallower. As such, these wetlands are more heavily used by domestic stock, particularly where they provide the only drinking areas. They have thus suffered greater degradation, which could often have been avoided by providing alternative drinking sources and controlling access.

An important factor affecting the susceptibility of a wetland to erosional degradation is its hydrogeomorphological setting. Besides seepage slope settings, discussed above, streambank or riparian sites are also considered susceptible because they are usually steep and subject to high hydraulic energy. The most noticeable effects of grazing of streambanks are:

1. a change, reduction or elimination of stream bank vegetation (e.g. the seedlings of favoured tree species may be eaten resulting in even aged stands of aging trees [Johnson and Corothers, 1982]); and

2. a change in the stream channel morphology by widening and shallowing the channel or by accelerating stream channel incision, depending on the soils and substratum type (Aucutt, 1988).

Several studies, including those of Gunderson (1968), Dahlem (1979), Duff (1979) and Kauffman *et al.* (1983a), report degradation of stream banks as a result of use by domestic stock. However, Hayes (1978), Knight (1978), and Buckhouse *et al.* (1981) found that stream bank loss did not occur more frequently in grazed riparian areas than in ungrazed riparian areas.

Buckhouse *et al.* (1981) found no significant difference between loss of banks grazed at 25-30 Animal Unit Months (AUM) per Metre of Accessible Streambank (MAS) and ungrazed streambank. A stocking rate of 48-50 AUM per MAS did, however, show a significantly greater stream bank loss than ungrazed streambank. Buckhouse *et al.* (1981) suggest that there is a threshold response rate of streambank loss. This would obviously vary according to characteristics of the site, such as soil erodibilty, stream hydraulic energy and nature of the vegetation cover. Kauffman *et al.* (1983a) conclude that management plans need to be geared for each particular riparian ecosystem as responses from land use activities vary from stream to stream. These recommendations, which are also applicable to other wetland settings, emphasise the importance of recognizing the special management requirements of different wetland areas.

Impact on the soil is also affected by the type of animal and how the animals move (as already mentioned in the discussion on soil poaching).

3.2.2.3 Effect of grazing on nutrient cycling

Grazing is likely to have an important effect on the exchange of nutrients in wetlands. Some of the organic carbon and nutrients consumed by domestic stock is removed as secondary production (in the form of harvested milk and meat). However, a large proportion of the carbon consumed is returned to the atmosphere through animal respiration and more than 90% of the consumed nutrients are returned as urine and dung. The nutrients in urine are immediately available, and although those in dung are less available, the decay rate is usually considerably higher than that of standing dead litter (Perkins *et al.*, 1978; Jensen *et al.*, 1990). Thus, grazing stimulates the turnover of organic matter and plant nutrients. Nutrient cycling may be increased five- to tenfold in some instances, which increases the exposure of these nutrients to leaching. Thus, it will be appreciated that stock grazing may detract from the water purification value of wetlands.

While domestic stock often graze lower, wetter wetland areas, they tend to rest and ruminate in the higher, less wet areas. Thus, their urine and dung deposition tends to be concentrated around the higher areas. This behaviour pattern contributes to redistribution and transport of nutrients from the lower parts of the wetland to the upper less wet parts.

3.3 Mowing of wetlands

Mowing has a similar effect to grazing in that it involves removal of aerial plant parts. As with trampling, the movement of harvesting machinery (usually a tractor) may disrupt and/or compact the soil. However, mowing differs from grazing in the following respects:

1. herbage removal is more uniform and less selective;
2. harvesting takes only a short time; and
3. smaller quantities of nutrients are returned to the wetland (unless animals are fed hay while on the wetland).

Other management actions sometimes associated with hay production are drainage to facilitate access, and fertilizer application to increase production. By altering the hydrology, drainage would detract from the erosion control, hydrological and ecological values (see Section 4.1). Fertilizer application is likely to detract from the water purification function (see Section 4.1) and ecological function. Bakker (1989) showed that hay cutting, in association with fertilizer application, resulted in a lower plant species diversity than hay cutting alone. It is suggested that fertilizer application results in a masking of subtle abiotic differences (e.g. slight differences in ground water depths).

Bakker (1990) found that salt marsh which was mown for hay production had a lower plant species diversity than did grazed salt marsh. He attributes this to the uniform close turf that becomes established under mowing. This offers fewer micro-habitats than the grazed marsh which is more heterogeneously defoliated. Nevertheless, wetland mowing has been widely shown to encourage greater plant species diversity than does unutilized wetland (Green, 1980; Bakker, 1990).

It appears that the timing of cutting is important. In relatively low producing (400 g dw.m^{-2}) wetland in the Netherlands, the site cut in autumn had a higher species richness than that cut in summer, whereas the reverse was true in high producing (800 g dw.m^{-2}) wet meadow (Bakker, 1989). Summer cutting of a productive wet meadow in the UK also resulted in higher species diversity than non-use of the stand, but autumn cutting did not result in any significant change in species diversity (Rowell *et al.*, 1985). Bakker (1989) suggests that a large standing crop in summer (which would accumulate more rapidly in a high producing meadow) disadvantages many species. Oomes and Mooi (1981) found that in an *Arrhmatherion elatioris* dominated area in the Netherlands, it was primarily the lower growing species (e.g. *Plantago lanceolata*) that decreased under autumn cutting.

Bakker (1989) reviewed studies examining the effect of cutting frequency on plant species richness. Hay-making twice a year gave the highest species richness, followed by annual hay-making and then hay-making every second year. Abandoned (unutilized) areas gave the lowest species richness values.

Very little work has been conducted on the effect of hay cutting on wetland-dependent animals. However, it is likely that, depending on the extent and timing of mowing, animals requiring vegetation cover would be disadvantaged by the immediate effects of cutting, particularly if it occurred during breeding. Bryan and Best (1991), observing that birds were most abundant in grassed waterways, in Iowa, USA, in July, recommended that mowing should not occur until the end of August.

3.4 Burning of wetlands

3.4.1 Reasons why wetlands are burnt

Schmulzer and Hinkle (1992) cite a number of authors (e.g. Viosca, 1931; Loveless, 1959; Cohen, 1974) to show that in many wetland systems, fires have occurred independent of human influence. This is supported by the observation that many wetland plants are relatively fire-tolerant (Loveless, 1959). Prior to anthropogenic fires, lightning is considered to have been the most important cause of wetland fires.

Fire is recognized as an important driving variable in wetlands and is used widely as a tool for wildlife management (Lynch, 1941; Schlictemeir, 1967; Ward, 1968; Smith and Kadlec, 1985; Mallick and Wein, 1986) and for enhancing stock grazing value (Lynch, 1941; Begg, 1990; Kotze *et al.*, 1994a). Where wetland areas pose fire hazards, controlled burns are used to remove the risk of runaway fires (Kotze *et al.*, 1994b). Some wetland grasslands are burnt to maintain the grass in a healthy state and to assist in alien plant control (Kotze *et al.*, 1994b; Otter, 1992). Other wetlands are burnt simply because they occur in frequently burnt landscapes, such as in the Highland Sourveld, and are not considered to warrant special protection by managers.

3.4.2 Effects of sub-surface fires on wetland functional values

Two broad types of fire occur in wetlands: surface and sub-surface fires. In surface fires, which are the most common, only the above-ground plant parts are combusted. Sub-surface fires, which are less frequent but more severe, consume above- and below-ground plant parts as well as soil organic matter. Sub-surface fires usually occur in soils rich in organic matter and are often referred to as peat fires (Ellery *et al.*, 1989). Organic soils tend to be permanently to semi-permanently flooded or saturated. Thus, they are usually only susceptible to sub-surface fires under very dry conditions (e.g. in a drought) or if the water regime has been modified to make them less wet (e.g. through drainage).

In surface fires, wetland plants usually rapidly re-establish vegetatively from the undamaged belowground parts and little physical change in the vegetation or soil results. In contrast, dramatic changes in soil and vegetation may result from sub-surface fires (Lynch, 1941; Cypret, 1961; Tallis 1983; Ellery *et al.*, 1989; Kotze 1992a). In spartina marsh, sub-surface fires kill *Spartina* spp. plants, which make up the dense climax vegetation, allowing more desirable food-plants such as *Eleocharis* spp., to increase in abundance, thus favouring muskrat and waterfowl (Lynch, 1941). Hydrological conditions at the time of the burn may have an important influence on the effect of fire. Mallik and Wein (1985) showed that burning when the water table had been lowered was more effective in decreasing the dominance of *Typha* than burning under high water table conditions, presumably because the fires penetrated deeper into the soil.

In the Okefenoke swamps, Cypret (1961) ascribes open water areas to the destruction of peat by fire. Cypret (1961) found that there was no relationship between the presence of lakes and the topography of the underlying sand floor. In other words, the lakes are holes in the peat rather than being depressions in the underlying sand, giving credence to the belief that the lakes were caused by fire (Cypret, 1961) A similar situation exists in the Wakkerstroom Vlei, Transvaal, where open water areas in the reed marsh consist of holes in the upper soft unconsolidated soil layers rather than depressions in the hard consolidated clay floor. Some of the open water areas in the reed-marsh present in 1991, resulted from a sub-surface fire in the drought of 1983 (Kotze, 1992a). Reed marsh soils in this wetland are not true peats, because they comprise only about 10% organic carbon, but have a high abundance of combustible roots. A comparison of airphotos from a sequence of Wakkerstroom Vlei dating from 1938 to 1990, shows that reed-marsh open water patches created by fire are recolonized by vegetation similar to the original within 20 years (Kotze, 1992a). This is considerably quicker than in certain peatland areas, where it may take between 100 to 300 years for the original vegetation to re-establish after the upper peat layers have been destroyed (Knight, 1991).

In the Okavango Delta, channel abandonment (caused by sediment deposition and vegetation blockages) results in desiccation of peat that formed under papyrus swamp in the channels. Sub-surface fires (which rapidly release retained nutrients) facilitate the conversion of this declining permanent swamp to a seasonally inundated flood plain or mixed terrestrial/aquatic habitat (Ellery *et al*, 1989). As such, Ellery *et al.* (1989) suggest that these fires contribute to the maintenance of habitat diversity and the overall structure of the Okovango Delta. Thus, from an ecological point of view, infrequent, localized sub-surface fires appear to be generally favourable in that they enhance habitat diversity.

Sub-surface fire may, however, detract from the hydrological and erosion control values of wetlands as it:

1. destroys organic matter and disrupts soil structure, rendering the soil more susceptible to erosion and decreasing the water storage volume of the soil;

2. releases trapped nutrients; and

3. destroys emergent vegetation.

In KwaZulu/Natal, the negative effects of sub-surface fires appear to be most pronounced in wetlands on seepage slope settings due to their small size, steep gradients and shallow soils. Recovery of the vegetation at these sites appears to be very slow, particularly when the soil has burnt down to the bedrock. However, where sub-surface fires cover a small proportion of a wetland, as usually occurs in large wetlands with deep soils, and in flat or depression settings, the overall impact on the system is likely to be negligible. De Beneditti *et al.* (1984) showed that revegetation of areas of depression wetland burnt by ground fires occurred within two years. However, had the ground fires occurred on a slope, these authors suggest that this might have had a severe impact and that the vegetation would have been considerably slower in re-establishing.

Lynch (1941) observed that when peat accumulating *Panicum* and *Spartina* marshes are left unburnt for several years, plant litter accumulates, resulting in a much deeper mulch on the soil surface. The marsh plants then produce roots in this layer, reducing root production in the deeper root horizons. The thick mulch layer makes these areas more prone to sub-surface fires and the change in root distribution renders the plants more susceptible to fire damage when such fires occur (Lynch, 1941). However, this phenomenon has yet to be quantified and would differ according to wetland type.

In summary, although sub-surface fires may enhance the ecological value of a wetland, they may also substantially detract from the wetland's hydrological and ecological values. The ultimate effect varies according to wetland type and conditions at the time of the burn. If sub-surface fires are considered undesirable, burning would have to be avoided in drought years, particularly at the end of the dry season, when soils are at their driest and are most susceptible to combustion. Very little work, other than that of Ellery *et al.* (1989), has been conducted on sub-surface fires in wetlands and the remainder of the discussion will deal with surface fires.

3.4.3 Effects of surface fires on hydrological and erosion control values of wetlands

The immediate effect of surface fires is the combustion of above-ground plant material with the loss of carbon and nitrogen to the air and the deposition of phosphorus and other minerals in the ash. Ninety per cent of the nitrogen from combustible plant materials was shown to be lost as a result of volatilization in both *Juncus roemerianus* and *Spartina cynosuroides* dominated marshes (Faulkner and De La Cruz, 1982). In tropical swamps and marshes where nutrient inputs are small and productivity is maintained by efficient internal cycling, nutrient loss during combustion may, in fact, lead to a reduction in primary production (Thompson, 1976; Whitlow, 1985).

Begg (1990), citing Downing (1966), Whitlow (1985) and Thompson and Shay (1985), states that there is evidence to suggest that indiscriminate burning of wetlands can be harmful to the water storage function of wetlands. None of the papers provide conclusive evidence to support this statement. However, it is fairly certain that burning of wetlands which are characterized by dry season die-back of above-ground plant material could be harmful to the water storage function of wetlands. Donkin *et al.* (1993) showed that evapo-transpirative loss of water during winter from wetlands with abundant standing dead material is less than the evaporative loss from open water. These wetlands, particularly in marsh areas, generally produce large amounts of standing dead material of a high reflectivity, the removal of which would promote evaporative loss from the wetland as surface litter results in a reduction in evaporative soil moisture loss. This is because the more exposed, or ash covered, soil or water surface absorbs considerably more solar radiation than the surface of an unburnt wetland. It is also more exposed to the desiccating action of wind.

A late winter/early spring burn would leave the wetland exposed for the shortest period. Thus, from a water conservation point of view, it would be preferable to an early winter burn, where the exposure period would be far longer.

In temporarily wet wetlands, the upper soil layers dry out during the dry season, irrespective of whether burning occurs or not. Since the hydraulic properties of the soil limit the movement of water to evaporating sites near the surface once the surface soil layers are dry, early winter burns are unlikely to affect the water storage function of temporary wetlands significantly. Thus, timing of burning is less important than in permanent wetlands.

As removal of standing dead material allows greater heating of the soil and improved light conditions for photosynthesising tissue, burning (as well as grazing) enhances early spring growth. This, in turn, promotes transpirative water loss from the wetland, but the effect would not persist for more than a few weeks. Sharrow and Wright (1977) found that herbage production in the early growing season was considerably higher where litter had been removed by fire. Similarly, Kauffman *et al.* (1983b) found that in ungrazed wet meadow communities, onset of the first season's growth occurred two weeks after that in the grazed wet meadow communities.

There are few studies on changes in soil nutrients in wetlands following fire other than those of Faulkner and de la Cruz (1982), Wilbur and Christenson (1983) and Schmalzer and Hinkle (1992). The latter found that:

1. soil pH increased immediately postburn but returned to preburn levels in 1 month;

2. organic matter increased in the first month, remained elevated for 9 months, then returned to pre-burn levels;

3. Ca, Mg, K and phosphate all increased in the first month, and the increases persisted for 6 to 12 months; and

4. ammonium-nitrogen and nitrate-nitrogen levels remained the same as the unburnt treatment but ammonium-nitrogen increased six months after the burn and nitrate-nitrogen increased 12 months after the burn.

These results are in general agreement with those of Faulkner and de la Cruz (1982) except that the latter did not measure organic matter. In both studies the soils were flooded at the time of the burn.

It is commonly held that fire, by reducing the input of organic matter into the soil via litterfall, decreases the organic matter content of wetland soils (Downing, 1966). In so doing, fires reduce the effectiveness of all those functions associated with a high soil organic matter content. However, Seastedt and Ramundo (1990), working on mesic grasslands, found that fires often do not lead to reduction in soil organic matter. This is because the removal of standing dead material (which would otherwise shade actively growing material) increases carbon fixation by photosynthesising tissue, resulting in increased root production. Thus, although input via litterfall is reduced, this is offset by enhanced rates of root detritus production.

It would appear that a similar situation holds for reed marsh. Both Mook and van der Troon (1982) and Thompson and Shay (1985) showed that belowground production by *Phragmites australis* was stimulated by spring and autumn burning. This is further supported by the positive relationship between burning and levels of organic carbon in marsh soils found by Schmalze and Hinkle (1992). Although it appears the wetland burning often does not decrease the organic carbon content of the soil, this may not be true for soils that are not flooded or saturated at the time of burning, particularly if sub-surface burning occurs.

Because of nitrogen volatilization losses during burning and the higher rates of carbon fixation, burnt grasslands tend to have roots with lower nitrogen contents than in unburnt grasslands. This enhances the immobilization potential of the soil, resulting in a decrease in the leaching of nitrogen (Seastedt and Ramundo, 1990). It thus appears that burning generally enhances the capacity of wetlands for removing nitrogen. However, at present this is speculative as it has not been investigated in wetlands. Schmalze and Hinkle (1992) note that soil nitrogen changes are very different from those observed in many non-wetland systems, because they are affected by seasonally varying water tables as well as by fire. The volatilization loss of phosphorus is considerably less than that of nitrogen, and the effect of fire on the capacity of wetlands for removing this element is likely to be even more difficult to predict.

3.4.4 Effects of surface fires on the ecological value of wetlands

3.4.4.1 Effects on wetland-dependent animals

Animal species populations may respond either positively or negatively to fire, or may show no response at all. Population responses are the result of direct or indirect effects of fire on individuals. Direct effects include increased mortality (induced by heat and asphyxiation), forced emigration, and reduced reproductive effort (Kauffman *et al.*, 1990). Bigham *et al.* (1965) and Vogl (1973) report minimal direct mortalities of birds and mammals associated with fire. However, although adult individuals of most wetland-dependent bird and mammal species are able to escape the direct effects of fire, juveniles may be far more vulnerable. This applies to above-ground nesting rodents and birds, particularly winter breeding birds such as wattled crane (*Grus carunculata*). In KwaZulu/Natal, fire has been shown to be the most important known cause of wattled crane chick mortality (Johnson and Barnes, 1991).

Indirect effects of burning may result from changes in quality and quantity of food and cover, availability of nest sites, predation pressure, intensity of competitive interactions, and patterns of social interactions. Ultimately, direct and indirect affects of fire on individuals lead to shifts in population density through time as micro-environmental conditions recover to their pre-fire status in the absence of further fire (Kauffman *et al.*, 1990). The snail *Neritina usnea* is more abundant the year following a fire, while ducks using *Juncus* marsh for nesting were found to prefer marsh burnt at least three years previously (Hackney and De la Cruz, 1981).

Fire positive species will tend to reach maximum levels in a matter of months following the fire, depending, of course, on the timing of the burn. In contrast, fire negative species may do so only several years after the fire. Clearly then, if the fire return frequency is considerably shorter than the recovery period of these species then the long term viability of their populations may be low. Little work has been done in KwaZulu/Natal, or internationally for that matter, on the recovery of wetland-dependent species populations following fire. While further studies may show otherwise, no wetland-dependent species in KwaZulu/Natal have been shown to have a recovery period longer than two years. Generally speaking, in the case of small mammals in the grasslands of the Highland Sourveld, a drastic decrease in the number of individuals occurs immediately after fire, followed by a rapid recovery, with numbers reaching pre-fire densities in 6 to 15 months (Rowe-Rowe and Lowry, 1982).

A comparison by Bowland (1990) showed the proportional species composition and density of small mammals in a frequently burnt wetland grazed by domestic stock to be similar to that of an infrequently burnt and ungrazed wetland. A controlled biennial burning programme, as opposed to complete protection, was shown to increase the breeding habitat value of sedge marsh for red-chested flufftail (*Sarothrura rufa*) (Taylor, 1994).

It should be noted, however, that the recovery rate may be strongly dependent on the existence of unburnt refuges (resulting from a patch/partial burn or a block burn) (Johnson, 1991) (Plate 11). The benefits derived by the presence of unburnt areas may extend well into the early growing season. This is demonstrated by the fact that for the first two months following partial burning of different marshes in KwaZulu/Natal, red-chested flufftail foraged in the burnt areas but always returned to the unburnt areas for shelter. Only after approximately two and a half months, when the vegetation cover had increased sufficiently, did they remain in the burnt areas (Taylor, 1994). It is presumed, therefore, that during the early growing season the adjacent unburnt areas allowed the birds to make greater use of the burnt areas. Thus, it appears that provided that wetland-dependent animals have adequate unburnt refuges from which they may recolonize burnt areas, then frequent (biennial) burning of wetlands in KwaZulu/Natal is unlikely to detract significantly from their ecological value. However, it is important to note that almost no work has been done in South Africa on the response of individual species to fire. Thus, some species may be favoured by a lower fire frequency. In order to account for the fact that species vary according to their preference for different stages of post-fire wetland recovery, an entire wetland, or group of closely situated wetlands, should not be burnt at one time. Instead burning should be in portions (burning blocks) such that at any given time a range of post-fire recovery stages are represented.

The immediate effect of burning appears to enhance the feeding potential of wetlands for certain species. Wattled crane numbers, for example, increased three-fold at Mgeni Vlei for the few weeks following its burning, as birds from the surrounding areas congregated at the burnt wetland (Kotze *et al.*, 1994c). Vogl (1973) found that alligators and 29 out of 35 bird species made greater use of a burnt pond shoreline than an unburnt shoreline. He suggested that removal of the dense mat of tangled flattened stems and leaves averaging 37 cm in depth that covered the water and soil allowed greater foraging efficiency by vertebrate species. Lynch (1941) found geese in a Louisiana marsh to exhibit a strong preference for burnt areas, where the abundance and availability of food was higher. Cypret (1961) suggested that the sandhill crane (*Grus canadensis*) population in the Okefenoke swamp may have been advantaged by intense fires that occurred in the 1954 and 1955 extended drought which caused an increase in some of their favoured foods. Smith and Kadlec (1985) found that waterfowl and muskrats preferentially graze burnt *Typha* and *Scirpus* marshes. This increased value may be due to:

1. grazers selecting for the increased nutritive quality of marsh plants, that has been shown to occur following burning (Faulkner and de la Cruz, 1982; Smith *et al.*, 1984); or

2. the absence of standing dead plant material that interferes with grazing.

Timing of burning may also have a profound influence on certain wetland-dependent species. Burning in early winter is likely to adversely affect winter breeding species such as the marsh owl (*Asio capensis*), while summer burning may adversely affect the many summer breeding species such as the purple heron (*Ardea purpurea*). Late winter/early spring burning is least likely to adversely affect breeding animals, as the majority of winter-breeders have completed breeding and the summer-breeders have yet to begin.

3.4.4.2 Effects on wetland-dependent plants

Fires are an important factor modifying the plant species composition and structure of wetlands. For example, marsh fires have been shown to be useful in sustaining desirable members of the Cyperaceae and Juncaceae (Vogl, 1974). Nevertheless, little work has been done on the long term effect of fire on plant species composition. An important indirect effect of fire on plants is the removal of loose surface and standing dead material, which favours the growth of new plant material by emergent herbaceous plants (Plate 12). Above-ground biomass production, inflorescence density and plant height at anthesis were found to be significantly greater in *Spartina pectinata* wetland which was burnt annually than in that which was burnt biennially (Johnson and Knapp, 1993).

Burning has been generally shown to prevent the invasion of herbaceous communities by woody plants, which are generally less resistant to fire. For example, prior to the exclusion of fire from the experimental catchment 9 in the KwaZulu/Natal Drakensberg, Cathedral Peak, the wetland area in this catchment consisted of several plant communities. These were the *Scirpus costatus, Oenothera rosea, Eleocharis dregeana* herbaceous communities and the *Leucosidea sericea* woody community (Killick, 1961). After 20 years of fire exclusion, this same area was described by Granger (1976) as comprising a single *L. sericea* community. In this case, fire enabled the wetland to support a far greater diversity of species and communities. It can thus be concluded that fire contributed positively to enhancing the ecological value of the wetland in catchment 9.

While most wetland plant species are well adapted to the direct effects of fire, they vary in their relative responses. The above-ground portions of certain wetland plants (e.g. *Juncus roemerianus*) often live for more than a year, in which case a surface winter fire would destroy living tissue. In contrast, species characterized by complete die-off of the above ground parts at the end of the growing season (e.g. *Phragmites australis*), do not lose any living tissue as a result of surface winter fires. Thus, one would expect that where these groups of species occur together, frequent winter fires would favour those species characterized by winter die-back. Conversely, wetland communities dominated by plants with long-lived aerial portions are less likely to change. This is partly because these communities cannot be burnt frequently and so will also resist changes in plant community structure. Hackney and de la Cruz (1981) found it very difficult to burn *Juncus roemerianus* marsh one year after a fire as there was insufficient combustible material.

The effect of burning on wetland plant communities is partly dependent on the timing of the burns, primarily through its effect on the dominant species. Spring burning, for example, enhances the performance of *P. australis*, as indicated by higher aerial and below-ground biomass and flowering shoot density (Mook and van der Troon, 1982; Thompson and Shay, 1985). In contrast, summer burns lowered the performance of *P. australis*, suggesting that summer burning has the potential for thinning dense reed stands and enhancing plant species diversity. Autumn burning appears to have an intermediate effect, resulting in higher biomass but reduced flowering shoot density (Thompson and Shay, 1985). Thus, where *P. australis* occurs as the dominant species in a mixed community, with other species such as *Molinia caerulea* and *Cladium mariscus*, burning to favour *P. australis* is likely to disadvantage or not affect the other species (Haslam, 1971), thereby lowering plant species diversity. Conversely, where burns disadvantage *P. australis*, burning may enhance plant species diversity.

The hydrological conditions, besides being important at the time of the burn, may also be important during the period following the fire. For example, increased mortality of sawgrass (*Cladium jamaicense*) results from flooding following fire (Lynch, 1951; Herndon *et al.*, 1991). It is suggested that the plants are most vulnerable to oxygen shortage resulting from flooding of their leaves immediately after burning because this is when their leaves are shortest (Herndon *et al.*, 1991).

In a comparison of a burnt and unburnt area of Nylsvlei, Otter (1992) found that in the burnt area, the abundance of *Themeda triandra* and *Oryza longistaminata* (both valuable grazing species) was greater and the abundance of *Asclepias fruticosa* (an alien weedy species) was considerably lower. Thus, these results suggest that in Nylsvlei, fire can be used effectively to control undesirable weedy species and promote the cover of desirable species for grazing animals without any obvious detrimental effects. Personal observation (1993) of comparable burnt and unburnt areas in Memel vlei, and Natabamhlope vlei, also indicate similar beneficial effects of fire.

3.5 Damming of wetlands

Wetlands are usually characterized by an impermeable foundation or obstruction (often a dolerite dyke) and a gentle upstream gradient -the very conditions sought by engineers for dams (Nanni, 1970). Consequently, in South Africa, where natural open water areas are scarce, numerous wetlands have been inundated by dams.

While dams are able to perform certain of the functions carried out by wetlands (e.g. sediment trapping and water storage) a dam is a poor substitute in certain respects (Begg, 1986). For example, the deepwater habitat that a dam provides for fauna and flora is very different from that previously offered by the now inundated wetland. Dams will often appear to be beneficial to the wildlife of the area in that this new habitat may attract wildfowl, such as Egyptian geese (*Alopochen aegyptiacus*). However, many of these are generalist species whose breeding and feeding areas are not threatened. In contrast, the habitat required by specialist wetland-dependent species is frequently lost. Other wetland functions that accrue from their shallow nature, such as the photodegradation of certain organic pollutants and the high degree of exchange between wetland water and sediment, would also be detrimentally affected.

The characteristic vegetation of wetlands lost when inundated by a dam, may be partly compensated for by that which develops around the shoreline of the dam, particularly at the upstream end where surface water tends to be shallower. Wetland vegetation development also commonly occurs below the dam wall as the seepage through farm dam walls is frequently high. Although these vegetation developments may provide some habitats resembling the previous wetland habitats, by no means do they usually replace the lost vegetation. Seasonal drawdowns and wave action often result in armoured barren shorelines (e.g. Hendrick Verwoerd Dam) which provide very poor habitat (Bruwer and Ashton, 1989). Seen on a landscape level, dam walls may obstruct the movement of aquatic animals, most notably fish. This applies particularly to dams that lack adequate fish ladders and result in periodic dry-season cessation of flow.

Large numbers of small farm dams (having walls <5 m high) have been built on virtually every river in South Africa (Noble and Hemens, 1978). Being small, they are often mistakenly thought to have very little effect on downstream flow. While the influence of an individual small dam may indeed be negligible during periods of high flow, this is seldom the case during low dry-season flows, particularly if water extraction is occurring from the dam. Where a series of dams are built along a river, the overall effect is compounded and can lead to the complete cessation of dry season flows (Bruwer and Ashton, 1989). In the Letaba River, for example, high rates of extraction from the numerous dams on the river have effectively transformed it from a perennial to a seasonal river. Dams can, however, have the opposite effect on dry season flows. Perenniality is enhanced where water extraction is low and adequate outflow is facilitated through the dam wall outlet and/or from seepage through the dam wall.

However, irrespective of whether dams increase or decrease dry season flow, one of their most adverse effects is on the first wet season flows. During the dry season, when river flows are reduced or cease (if the river is seasonal) the levels of most dams drop through evaporation and/or abstraction. This results in the first wet season flows being retained until the dam is sufficiently full. This can cause considerable alteration in the timing, and thus, success of the life cycle stages in the river biota, as well as negatively affecting human users downstream (Bruwer and Ashton, 1989). Dams may also negatively affect downstream biota by changing water temperatures, oxygen levels, silt loads and ionic concentrations (Davies and Day, 1986). An additional disadvantage of dams occurs in the frequent case where they burst after a heavy rainfall contributing to increased flood damage and sediment release.

It has often been perceived that wetlands "waste" water as a result of their associated vegetation "pumping" water into the atmosphere through transpiration, and that local water resources could be improved by flooding the wetland area permanently with a storage dam. This view arose largely out of the earliest investigations of the evapotranspirative losses from a marsh compared with evaporative losses from open water. These studies (e.g Blaney and Ewing, 1946) reported losses to be higher from marsh areas. However, the results of these early investigations have since been called to question. Those of Blaney and Ewing (1946, as cited by Linacre *et al.*, 1970), for example, were calculated using the Blaney formula for evaporation (Blaney, 1952), which ignores the effect of humidity and wind variations. In addition, the study assumed the applicability of empirical coefficients derived from measurements 500km away (Linacre *et al.*, 1970). More recent studies (Eisenlohr, 1966; Paijmans, 1985; Chapman, 1990) have reported losses from vegetated wetlands to be similar to or lower than from open water. A number of factors contribute to this, such as the high reflectivity of the plant canopy and the shelter it provides to the water surface against wind (Linacre *et al.*, 1970).

Linacre *et al.* (1970), in a general summary, state that in dry climates wetlands usually lower evaporation, and in wet climates, while this also often occurs, the likelihood of their enhancing evaporation is higher. However, such generalizations are dangerous: there are numerous factors affecting evapotranspiration, such as solar radiation inputs, wind, surface water depth and whether the plants are vigorously growing or are dormant. In conclusion, it can safely be said that when wetland plants are not actively growing and transpiring (i.e. when die-back has occurred) water loss would be lower than that occurring from a comparable open water area.

4. CONCLUSION

In conclusion, this review demonstrates clearly that wetlands possess numerous functional values which may be of great value to society. Many of these are not readily apparent and are easily overlooked. Wetland functional values range from those which have a geographically defined service area from which benefits are potentially derived (e.g. flood attenuation) to functions which do not have geographic limits but rather have a global influence (e.g. biogeochemical cycling).

A very large body of information exists concerning wetland functional values and the effect of different land-uses on these. Much of what is known about wetland functional values is the result of short-term research projects examining a single process in one geographic location. Hence, great uncertainty is often involved in extrapolating from these studies. For example, the extent to which a wetland is trapping pollutants is not only dependent on the nature of the wetland but also on the pollutants involved. Thus, it is very difficult to predict how a given wetland is likely to carry out this function and even more difficult to quantify how this is likely to be affected by different land-uses. Nevertheless, general principles relating to the nature of wetlands and determinants of wetland structure and function, allow for qualitative predictions.

From the discussion on wetland functional values it can be seen that the hydrological regime (encompassing such factors as frequency and duration of flooding and hydrological energy) is the most important factor directly influencing physical and chemical processes in a wetland (e.g. degree of substrate anoxia, nutrient retention and sedimentation patterns). These influences on the physical and chemical environment, in turn, have a direct effect on the wetland biota. Clearly, the hydrological regime is the principal factor affecting the functional values of wetlands and it is useful to view all impacts in terms of their effect on the hydrological regime. Alterations to wetlands (arising out of different land-uses) can be reduced to two main groups of actions:

1. those that directly change the hydrological regime as a result of substrate disturbance (e.g. hoof action, tillage, and construction of drainage channels); and

2. those that remove plant material (harvesting, grazing and fire). Because of the influence of vegetation on wetland hydrology, removal and disturbance of wetland vegetation also has the potential to influence the hydrological regime.

In attempting to predict the impact of a given land use, it may be assumed that the greater the extent to which the hydrological regime is disrupted, the greater will be the impact. This review also focuses on the need for attention to additional features, including:

1. susceptibility to erosion (determined by *inter alia*: soil erodibility, hydrogeomorphological setting, slope and climate);

2. habitat value for wetland-dependent species (the greater the number of valued wetland-dependent species supported by a given wetland, the greater will be the likelihood of a loss of ecological value if the wetland is developed); and

3. extent and historical loss of wetlands in the surrounding landscape. Seen in a landscape context, the loss of functional values in a given wetland is considered to have a greater impact if a large proportion of the wetlands in the surrounding landscape had already been lost than if a small proportion had been lost.

Land-uses vary greatly with regard to the effect they have on wetland functional values (Table 3). Drainage and the production of crops represents the severest form of disruption, involving the permanent removal of the native wetland vegetation, a lowering of the water volume and retention time in the wetland and regular disturbance and exposure of the soil. Of the land-uses discussed, pasture production is second in severity. Annual pasture species having a low wetness tolerance (e.g. *Lolim multiflorum*) have a more severe impact than perennial species with a higher wetness tolerance (e.g. *Festuca arundinacea*). Judiciously managed grazing of undeveloped wetlands is considered to be the least severe as it involves minimal disruption of the hydrological regime and does not involve the replacement of the native species. However, when there is mismanagement where wetland soils are of high erodibility, grazing has the potential to be equally, and in some cases more, disruptive. Heavy stocking rates lead to accelerated erosion caused directly by hoof action on the substrate, and indirectly through a reduction in the health of the wetland vegetation and a lowering of its ability to control erosion. This leads to the formation of gullies that lower the water table, as would occur in a wetland that had been intentionally drained.

Burning is not a land-use *per se* but is often used to enhance the grazing value of wetlands. Although the loss of wetland functional values due to burning may be substantial, it is usually small and in many cases burning may enhance such values. Dams perform certain wetland functions (e.g. sediment trapping) but are often poor substitutes for others such as the provision of habitat for wetland-dependent species.

As the demand for resources escalates because of the exponentially increasing human population, it will become increasingly unrealistic to call for the non-use of wetlands. In order to achieve a trade-off between maximising the benefits derived by different wetland users and minimizing the loss of functional values to society, it is important to understand how the different wetland functional values are affected by various land-uses. This review has concentrated on those agricultural land-uses commonly applied to wetlands in the midlands of KwaZulu/Natal (Bioclimatic regions 2, 4, 6 and 8, according to Phillips, 1973). Although it has focused on the types of wetlands found in this part of KwaZulu/Natal, the general principles dealt with are equally relevant to other regions and land-uses.

Table 3 Impacts of various land-uses on the flood attenuation, baseflow augmentation, water purification, erosion control and ecological values of wetlands as mediated through important characteristics that influence such values

A. VELOCITY REDUCTION	LAND-USE				
CHARACTERISTIC	Graze	Over graze	Past- ure	Crop	Dam
Surface area of active floodplain.	0	--	-/0	-/0	0
Surface roughness (vegetation and ground surface)	-/0	-	-	-	-
Slope	0	-	0	0	0
Detention storage capacity	0	--	-	--	+/0
Sinuosity of channels	0	-/0	-	-	NA
OVERALL IMPACT	0	-	-/0	-	0

B. FLOOD ATTENUATION	LAND-USE				
CHARACTERISTIC	Graze	Over graze	Past- ure	Crop	Dam
All characteristics influencing velocity reduction.	See functional value A.				
Soil saturation	0	0	+	+	-
OVERALL IMPACT	0	-	-/0	-	0

C. EROSION CONTROL	LAND-USES				
CHARACTERISTIC	Graze	Over graze	Past- ure	Crop	Dam
All characteristics influencing velocity reduction.	See functional value A				
Vegetation cover	0/-	--	-	--	-
Disturbance level	0				
OVERALL IMPACT	0/-	-	-	--/-	0

Table 3 continued

D. WATER PURIFICATION	LAND-USE				
CHARACTERISTIC	Graze	Over graze	Past-ure	Crop	Dam
All characteristics influencing velocity reduction	See Functional Value A				
All characteristics influencing erosion control.	See functional Value C				
Vegetation cover	0/-	0/-	-	-	0
Disturbance level	0	0	-	--	0
OVERALL IMPACT	0/-	0/-	-	--/-	0/-

E. HABITAT VALUE	LAND-USE				
CHARACTERISTIC	Graze	Over graze	Past-ure	Crop	Dam
All characteristics influencing velocity reduction.	See functional value A				
Native species replacement	0	-	--	--	-
Disturbance level	+	-	--	--	+/-
OVERALL IMPACT	+/-	-	--/-	--/-	-

LEGEND

+	Positive influence
++	Strong positive influence
-	Negative influence
--	Strong negative influence
+/-	Influence positive or negative but usually not strongly so in either direction
0/-	Influence negative or negligible

Graze: Stock grazing of natural wetland without gully erosion occurring.
Over-graze: injudicious grazing management leading to severe gully erosion. Injudicious management associated with pasture and crop production may also lead to severe gully erosion.

Pasture: Perennial pasture production. Annual pastures are best considered with crops due to lower wetness tolerance of the species and more frequent soil disturbance.
Crop: Crop production.
Dam: The assessment of dams is made on the assumption that they do not burst, which does not always hold.

Although velocity reduction *per se* is not generally considered a functional value it is included because it directly influences all other functional values.

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6. GLOSSARY

Bioclimatic Group: Phillips (1973) classified the extremely varied natural resources of Natal into 11 bioclimatic regions based primarily on climatic parameters. These groups provide convenient natural resource classes in terms of which management guidelines can be formulated.

Biological integrity: refers to the fauna and flora that are characteristic of an area (i.e. the naturalness of the area).

Ecological value: refers to the value of the wetland in maintaining the biotic diversity of the area. Biotic diversity can be measured at many different levels making it almost impossible to prescribe a standard method to describe it. Its assessment may be simplified by determining the degree to which management is affecting biological integrity and populations of valued species.

Hydrology: is the study of water, particularly the factors affecting its movement on land.

Impact site: that part of the wetland site to which a proposed land-use is to be applied.

Hydrogeomorphological setting: the landform setting (which influences surface water flow patterns within the wetland) and the position relative to other landforms in the wider landscape.

Marsh: Marsh is usually dominated by tall (usually >1.5m) emergent herbaceous vegetation, such as the common reed (*Phragmites australis*). It tends to be semi-permanently or permanently wet.

***n* Value:** The *n* value refers to the relationship between the percentage of water under field conditions and the percentages of inorganic clay and humus and can be approximated in the field by a simple test of squeezing the soil in the hand. It is helpful in predicting the degree of subsidence that will occur after drainage and whether the soil may be grazed by livestock or will support other loads (Pons and Zonneveld, 1965; Soil Survey Staff, 1990).

Open water: Open water comprises temporarily to permanently flooded areas characterized by the absence (or low abundance) of emergent plants.

Permanently wet: The soil is flooded or waterlogged to the soil surface throughout the year, in most years.

Permanent wetland: A wetland with a permanent water regime.

Red Data species: Red data species refer to all those species included in the categories of endangered, vulnerable or rare, as defined by the International Union for the Conservation of Nature and Natural Resources (Smithers 1986).

Roughness coefficient: The roughness coefficient is an index of the roughness of a surface and is a reflection of the frictional resistance offered by the surface to water flow.

Seasonally wet: The soil is flooded or waterlogged to the soil surface for extended periods (>1 month) during the wet season, but is predominantly dry during the dry season.

Seasonal wetland: A wetland with a seasonal water regime.

Temporarily wet: The soil close to the soil surface (i.e. within 40 cm) is occasionally wet for periods > 2 weeks during the wet season in most years. However, it is seldom flooded or waterlogged at the surface for longer than a month.

Temporary wetland: A wetland with a temporary water regime.

Wet grassland: Wet grassland is usually temporarily wet and supports a mixture of: 1) plants which are common to non-wetland areas and 2) short (< 1m) hydrophytic plants (predominantly grasses) common to the wet meadow zone.

Wet meadow: Wet meadow is usually seasonally wet and is usually dominated by hydrophytic sedges and grasses common to temporarily or seasonally wet areas.

Wetland: Land where an excess of water is the dominant factor determining the nature of the soil development and the types of plants and animals living at the soil surface (Cowardin *et al.*, 1976).

Wetland functional values: Where wetland functions (e.g. the trapping of sediment) are of value to society, they are termed functional values. Wetland functions refer to the many physical, chemical and biological processes that take place in wetlands.