ENVIRONMENT PROTECTION AUTHORITY

Consultancy report

Nutrient movement through SA soil

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Government of South Australia

EPA South Australia

Nutrient Movement Through SA Soil

Literature Review

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June 2004
Ref No 20040122RA01

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Document History and Status

Rev	Description	Author	Rev'd	App'd	Date
A	Draft for comment	MS	Jim Cox VAH	VAH	30/06/04

Acknowledgements

Tonkin Consulting would like to acknowledge and thank Dr Jim Cox from CSIRO for his input, support, ideas and review in the undertaking of the entirety of this project. Tonkin also acknowledges Dr Cox's photographs, which have been used on the front page of this report, with his kind permission.

Glossary of Terms

A1 Horizon	Mineral horizon at or near the soil surface which is darker in colour than the underlying layers due to accumulation of decomposed organic matter and has maximum biological activity (McDonald and Isbell, 1990).
Adsorption	The interaction of ions with the surfaces of soil materials, particularly organic matter and clay (Charman, 1991).
Aerobic	Soil condition where free oxygen is plentiful and the soil chemical reactions are predominantly oxidation reactions. This condition is usually found in well-structured, freely-draining soil (Charman, 1991).
Ammonification	Processes which transform soil nitrate to soil ammonium.
Anaerobic	Soil condition in which free oxygen is deficient and the soil chemical reactions are predominantly reduction reactions. These conditions are found in waterlogged or poorly drained soil where the water had replaced air in the soil pores (Charman, 1991).
B2 horizon	Mineral horizon in which the dominant feature is one or more of the following: ▪ high concentration of silicate clay, iron, aluminium or humus either alone or in combination; ▪ maximum development of pedological organisation as evidenced by a different structure and/or consistence and/or stronger colours than the A horizons above or any horizon immediately below (McDonald and Isbell, 1990).
Bioturbation	The churning and stirring of sediment by organisms (Bates and Jackson, 1984).
Denitrification	Processes which transform soil nitrate into gases N2 and N2O (Strong and Mason, 1999).
Electrical Conductivity	Measure of the extent to which water or the soil solution conducts an electrical current, which is dependent upon the total and relative proportion of soluble salts (dissolved ionised substances) and the temperature (Rayment and Higginson, 1992).
Erosivity	The kinetic energy and intensity of a storm which translates into the ability of the rainfall to erode the soil surface (Elliott and Leys, 1991).
Erodibility	The susceptibility of a soil to the detachment and transport of soil particles by erosive agents (Charman, 1991)

Eutrophication	The process whereby a waterbody receives inputs of large quantities of nutrients resulting in high productivity of aquatic plants, such as algae (Bates and Jackson, 1984).
Evapotranspiration (ET)	The portion of precipitation that returns to the atmosphere from evaporation from soil and plant surfaces or is transpired by the plant (Bates and Jackson, 1984).
Field Capacity	The amount of water held in a soil after any excess has drained away by gravity following saturation. As a general rule, the soil is considered to be at field capacity after draining for 48 hours (Charman, 1991).
Heterotrophs	Micro-organisms that obtain their carbon to make cellular materials from organic compounds (Bowyer et al., 1988).
Immobilisation	Processes which transform soil mineral nitrogen, i.e. nitrate, nitrite or ammonium, into soil organic nitrogen (Strong and Mason, 1999).
Macropores	Approximately circular spaces or openings in the soil not occupied by mineral matter. The cross-sectional diameter is greater than 0.075 mm diameter and can be seen by the naked eye (McDonald and Isbell, 1990).
Mineralisation	Processes which transform soil organic nitrogen into mineral nitrogen, i.e. nitrate, nitrite or ammonium (Strong and Mason, 1999).
Nitrification	Processes which transform ammonium into nitrate, via nitrite (Strong and Mason, 1999).
Sodic	Where the soil contains sufficient exchangeable sodium to adversely affect soil stability, plant growth and/or land use, usually defined as where the soil has an exchangeable sodium concentration as a percentage of total cation exchange capacity (ESP) of 6%.
Sodium Absorption Ratio	The sodium absorption ratio (or SAR) is calculated as the ratio of sodium to the square root of half the sum of calcium plus magnesium. The SAR is used to indicate any potential hazard from irrigation waters to plant growth or soil structure or permeability (Rayment and Higginson, 1992).
Structure	Description of the distinctness, size and shape of soil aggregates (peds). Peds can be identified as a cluster of primary particles separated by a plane of weakness from other such clusters (McDonald and Isbell, 1990).
Texture	Description of the size distribution of individual mineral soil particles finer than 2 mm (McDonald and Isbell, 1990).
Volatilisation	Processes which transform soil ammonium into ammonia (NH_3) gas (Strong and Mason, 1999).

Abbreviations

μ	micro (10^{-6})	g	gram	N	nitrogen
m	milli (10^{-3})	t	tonne	P	phosphorus
c	centi (10^{-2})	gal	gallon	TOC	total organic carbon
d	deci (10^{-1})	ha	hectare	Al	aluminium
k	kilo (10^{3})	min	minute	Fe	iron
M	Mega (10^{6})	yr	year	Mn	manganese
		S	Seimens	EC	electrical conductivity
		mol	mole	SAR	sodium absorption ratio

1. Introduction

1.1 Background

The South Australian Environment Protection Authority (SA EPA) approves and comments on irrigation schemes, especially those involving effluent from households or industry.

Individual households may have on-site wastewater treatment systems or may be part of a Septic Tank Effluent Disposal System (STEDS). In these household schemes, a package treatment system is used to treat wastewater from toilets, washing machines, showers, etc. to an approved standard. The treated effluent is then disposed of via subsurface trenches or sprinklers to an irrigation area, which may or may not be a dedicated site. Subsurface trenches rely on the soil's ameliorative capacity and microbial population to utilise the applied nutrients. Sprinkler irrigation utilises plant requirements and soil ameliorative capacity to utilise the applied nutrients and water.

Wastewater Treatment Plants (WWTPs) accept effluent from household, commercial and industrial premises. The wastewater is treated by aerobic and/or anaerobic processes to an acceptable standard. The final effluent may be disposed of via irrigation to land or disposal to water bodies, eg. discharge to ocean or rivers. In coastal council areas, foreshore watering may be undertaken whereby effluent is applied to foreshore areas and allowed to leach through the underlying sand to sea. Irrigation schemes are based on full-utilisation of effluent, while foreshore watering only achieves partial utilisation of effluent by plants and soil microflora.

Agricultural and food processing industries, such as piggeries, abattoirs and wineries, produce effluent as part of their operations. This effluent may be disposed of to land via irrigation systems. The quality of the effluent varies between operations depending on their treatment system. There are no specific approved standard for treatment of effluent from these industries.

The concentrations of nutrients, particularly nitrogen and phosphorus, vary between the sources of effluent, depending on their source and treatment. The potential impacts on the environment depend upon the quality of the effluent, the loadings to the environment and the capacity of the environment to ameliorate the added loads. The capacity of the environment to ameliorate added loads depends on the receiving environment and alterations to the receiving environment may improve its capacity to ameliorate the wastewater applied. In addition, buffer zones or set back distances may be used to minimise the potential impact on nearby sensitive receptors.

SA EPA contracted Tonkin Consulting in April 2004 to investigate nutrient movement through South Australian soil and to report the findings of the investigation in a report to be used by EPA officers when assessing applications related to effluent disposal.

1.2 Scope of Works

The following tasks will be undertaken as a part of the work:

- Literature review to identify the major influences on nutrient movement in South Australian soil and strategies for minimising nutrient movement under irrigation in Australia and overseas;
- Review overseas and Australian guidelines and published literature for recommended set back distances and the rationale for setting these distances and compare these to the South Australian environment;
- Prepare a comprehensive document that is clear, concise and targeted towards use by EPA officers;
- Meet with EPA on two occasions to present project findings, to discuss project focus and to ensure EPA's desired outcomes will be met.

It is understood that the purpose of the report is to aid the EPA and its officers in assessing the nutrient leaching potential of proposed and current effluent irrigation operations in South Australia. Coastal areas were identified as a particular concern due to the practice of foreshore watering by local Councils and the proximity of septic absorption trenches to the intertidal areas.

2. Nutrient Movement Mechanisms

Nutrient movement through soil is predominantly controlled by water movement through soil. Some nutrient translocation may occur through bioturbation, the mixing of the soil by soil fauna such as ants and worms, however, this is unlikely to be the major mechanism for nutrient movement.

Soil formed in the temperate regions has relatively deep topsoil enriched with organic matter and is inherently fertile (Power, 1990). Soil formed in dryland agricultural areas in subtropical areas, is often older, more weathered and inherently less fertile. As a result of these differences, the movement of water and nutrients and the importance of various factors will vary between regions.

2.1 The Water Balance

Water is added to the soil through precipitation (including rainfall, snow, frost and dew) and irrigation and then stored in the soil and/or lost via evapotranspiration, runoff or drainage. The water balance that relates these factors is:

$$P + I = ET + R + D + \Delta S$$

Where,

P	is precipitation;
I	is applied irrigation;
ET	is evapotranspiration;
R	is runoff;
D	is drainage;
ΔS	is soil storage.

The applied water may run off the surface, infiltrate into the soil or evaporate off the soil and plant surfaces. Once the water has infiltrated into the soil it may be taken up by plants, which in turn transpire water returning it to the atmosphere, move laterally if there is a slope or an impeding subsurface layer (throughflow), or leach deeper into the soil. Eventually, leaching water, runoff and throughflow may reach surface water or groundwater. A simplified water cycle, excluding irrigation, is shown in Figure 2.1.

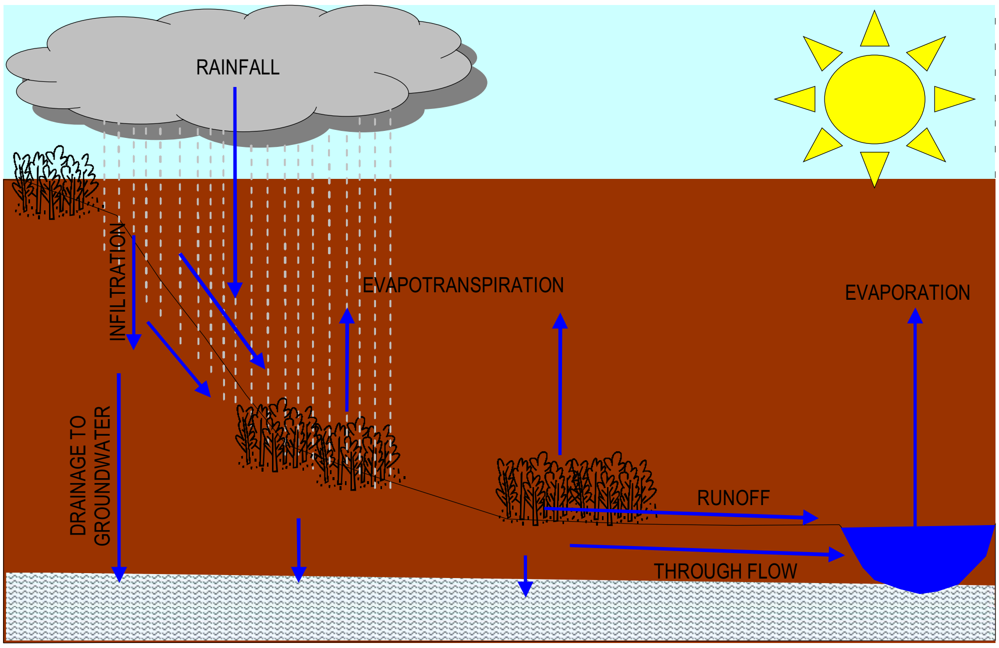

Figure 2.1 A Simplified Water Cycle

2.1.1 Soil Erosion

As water moves over or through the soil, erosion may occur. Erosion is described by three stages: detachment, transport and deposition. Detachment occurs once the shear stress exerted by raindrop impact or overland flow exceeds the cohesive force of soil particles and organic matter (Merritt *et al.*, 2003). Transport may be over or through the soil and deposition will begin to occur once the flow velocity and turbulence reduces. The Universal Soil Loss Equation (USLE) is an empirical model developed to predict long-term average soil loss (Elliott and Leys, 1991) and takes the form:

$$A = R \times K \times (L \times S) \times C \times P$$

Where:

A is the predicted long-term average soil loss (t/ha/yr);
R is a measure of erosivity of the rainfall and runoff;
K is a measure of the erodibility of the soil;
L , S are the slope length and slope steepness;
C is the cover management factor;
P is a land management factor.

From this simple model, it is evident that the climate, soil type, topography, plants and land management (eg. soil conservation works, ploughing) are all factors affecting the amount of sediment removed from a site.

2.1.2 Soil Infiltration and Drainage

Water infiltrates into the soil and then moves through the soil in response to a moisture gradient and may move in saturated or unsaturated conditions (Collis-George, 1988). The rate at which water moves through soil is a function of soil texture and structure. In particular, the presence of macropores may result in water "short-circuiting" the main soil bulk and transmitting water relatively rapidly to deeper in the profile than is predicted from the soil bulk properties.

Deep drainage results from water moving below the root zone of the plants growing on the site and hence can not be removed from the soil by evapotranspiration. This deep drainage may reach groundwater depending on the depth to groundwater and the hydrology of the soil and rock layers.

2.2 Nitrogen Cycle

Nitrogen (N) is a plant macronutrient and hence is an important input in agriculture. In addition, nitrogen is one of the major constituents of human and animal wastes, such as manure, biosolids and effluent. The Australian Bureau of Statistics (www.abs.gov.au) notes that the annual release of nitrogen from fixed sources is 350 Mt (1×10^{12} grams), which is comprised of 210 Mt from anthropogenic sources and 140 Mt from natural sources. Nitrogen released from fertiliser is 80 Mt/yr, half of which is not taken up by the plants but is evaporated or leached into surface and groundwater.

Nitrogen inputs to the soil are from atmospheric deposition, fixation, fertiliser, plant residues and animal urine and manure. Nitrogen is present in the soil as organic nitrogen, ammonium (NH_4^+) and nitrate (NO_3^-). Nitrite (NO_2^-) may also be found in soil but is an intermediate step in the conversion between ammonium and nitrate and is generally highly unstable. Losses of nitrogen are by gaseous loss, leaching and runoff and through removal of plant and animals products, eg. grain, hay, milk, meat and wool. Inputs and outputs for nitrogen can be represented by the nitrogen cycle (Figure 2.2).

In the UK and Europe, the estimated nitrogen flows within a dairy are to apply 300-400 kg N/ha/yr, 20% of the total nitrogen inputs (including that from fertiliser, imported feed, bedding etc.) are taken off the farm in milk and younger animals, 46% is lost through leaching, denitrification and volatilisation and 34% is unaccounted for (Jarvis, 1993). In subtropical Australia, crops remove 18-50% of N applied and a nitrogen tracer trial in NSW found after 6 months 25% of applied nitrogen was held in the plant, 25% in the soil and plant roots and 50% was lost due to volatilisation, denitrification and leaching (Weier, 1994).

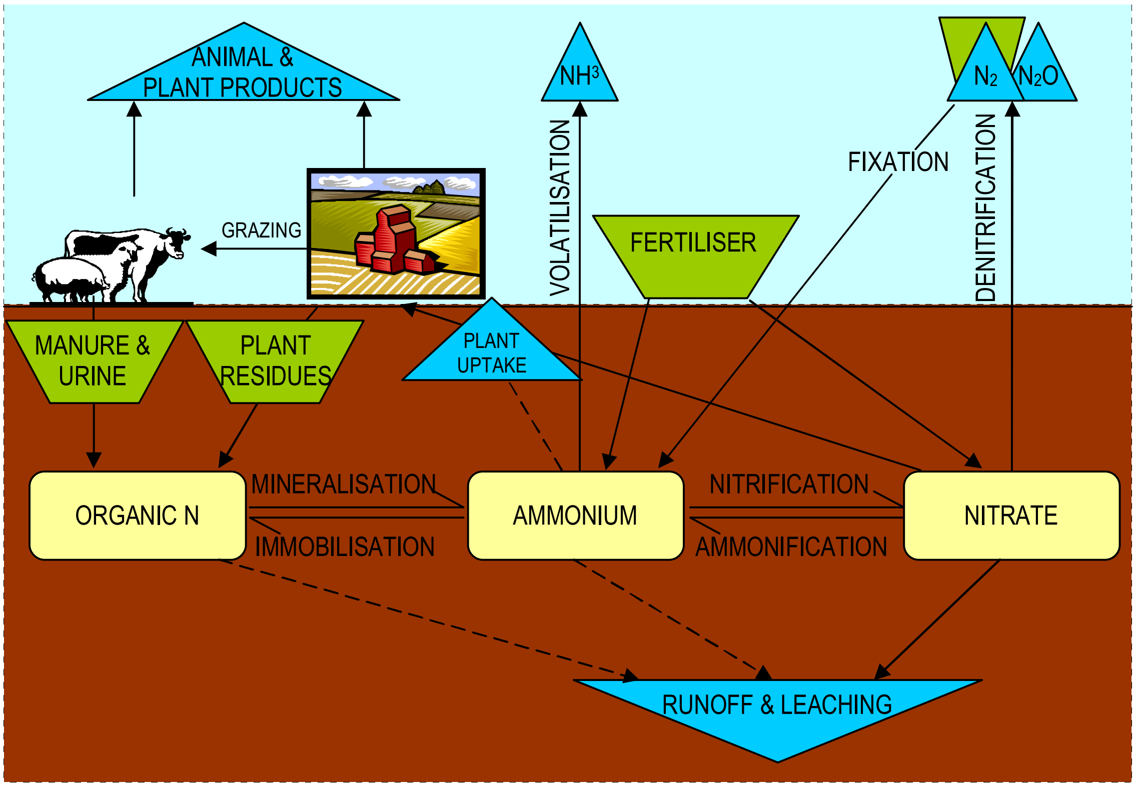

Figure 2.2 The Nitrogen Cycle showing inputs to ( ), transformations in ( ) and losses from ( ) the soil.

2.2.1 Organic Nitrogen

Nitrogen is present in the soil as organic forms, including attached to organic matter and within microbial cells, and mineral forms, i.e. nitrate and ammonium. Microbial decomposition of organic matter and microbial hydrolysis of proteins and amino acids converts organic N to ammonia and is known as mineralisation. Mineralisation is not undertaken by specific micro-organisms, however a strong relationship exists between total organic carbon and nitrogen mineralisation. When carbon sources are added to the soil, eg. hay or stubble, nitrogen “drawdown” may occur where micro-organisms immobilise nitrogen (Strong and Mason, 1999) while breaking down the added organic carbon. Hazelton and Murphy (1992) note that the C:N ratio in arable soil is usually 10-12 with ratios less than 10 indicating rapid decomposition, between 15-20 indicating slowed decomposition and ratios greater than 25 indicating organic matter is likely to remain “raw”.

Organic N is hydrolysed to ammonium when an energy source (carbon) is available to the heterotrophic micro-organisms.

2.2.2 Mineral Nitrogen

Mineral nitrogen includes ammonium, nitrite and nitrate, with nitrite and nitrate also referred to as oxidised nitrogen (or NO_x). Mineral nitrogen is available for uptake by plants.

Fixation of nitrogen gas and production of ammonium is undertaken by a range of free living and symbiotic bacteria. The best known nitrogen-fixing bacteria include *Rhizobium* and *Azotobacter*, which are associated with legume roots, and *Anabaena* and *Nostoc*, which are cyanobacteria (also known as blue-green algae).

Ammonium may then be oxidised by specialised aerobic bacteria, *Nitrosomas, Nitrococcus* and *Nitrobacter*, to nitrite and then nitrate. Ammonium is readily converted to nitrate in the presence of O_2 (required as an electron receptor). pH, temperature and soil moisture content also affect rate of nitrification. Linn and Doran (1984) showed that N mineralisation, nitrification and CO_2 production increased as the percentage of soil pores filled with water increased to 60% filled, i.e. approximately field capacity.

Ammonium is readily oxidised to nitrate in warm, moist, aerobic conditions by specialised bacteria.

Ammonification, the conversion of nitrate to ammonia, is undertaken by fungi as well as bacteria and can proceed in dry conditions due to greater fungal tolerance of dry conditions compared to bacteria. Nitrification is caused by bacterial processes only and hence ceases in dry conditions or other times when bacteria are not active.

Ammonium is a cation and hence can be held by the soil's cation exchange complex and is not readily leached. Davey (1988) notes that some clay minerals (such as smectites) in the soil can "lock up" ammonium due to the clay mineral collapsing and forming a different mineral structure (similar to mica), though this is not common in soil. Once the exchange capacity is "full", ammonium may leach through the soil.

Ammonium may be held in the soil on the cation exchange sites, taken up by plants or volatilise, depending on concentrations and moisture conditions.

Nitrate is an anion and hence is repelled by the clay minerals and not held on the exchange sites. Variably charged sites are present in organic matter and nitrate may be loosely bound. As a result of weak adsorption and fewer adsorption sites, nitrate is more easily leached through the soil than ammonium.

Nitrate is readily leached from the soil, taken up by plants or denitrified.

Gaseous loss of ammonia and nitrate can occur. The main pathway for loss of nitrogen as ammonium is through volatilisation, i.e. gaseous loss through conversion of NH_4^+ to NH_3 while nitrate is lost as nitrogen gas and nitrogen peroxide. Wet soil, near saturation, favours anaerobic processes such as denitrification (Linn and Doran, 1984). Alternate wetting and drying cycles also lead to denitrification losses by nitrification in the drying cycle and then denitrification during wetting cycles.

2.2.3 Influences on Nitrogen Speciation

Micro-organisms are the major influence on nitrogen speciation and hence factors that affect microbial survival and growth also affect nitrogen speciation. Duxbury and New (1988) discuss the microbial ecology of the bacteria, actinomycetes, fungi, algae, protozoa, flagellates, ciliates and amoebae in soil. Most micro-organisms associate with organic particles, which are sources of nutrients. Ridley *et al.* (2001) found that mineralisation of organic N occurred mainly in the A1 horizon, where the organic matter accumulates.

Soil moisture and temperature are also important factors which influence microbial growth and hence nitrogen speciation and loss. Ridley *et al*. (2001) measured increased nitrate concentration throughout the soil profile in spring due to leaching from the upper layers during winter. By late spring, nitrate and ammonium had begun to accumulate due to ammonification and nitrification of organic N in the warm, moist conditions.

Xu *et al.* (1996) measured nitrogen in soil profiles at 123 field sites in South Australia prior to sowing over three growing seasons (1990-1992). The distribution of nitrate and mineral N (nitrate, nitrite and ammonium) through the profile was similar at the field trial sites. Nitrate was found to be the predominant form of mineral N in the upper portion of the profile with ammonium concentrations more highly correlated to mineral N at depth. This was probably due to the lack of nitrification at depth from limited aeration and organic matter at depth.

2.2.4 Nitrogen Movement and Losses

Nitrogen loss from the soil may be as a gas, as a result of volatilisation of ammonia or denitrification of nitrate, or as a solute in runoff, throughflow or drainage. In general, high pH, low cation exchange capacity (CEC) and light texture contribute to greater volatilisation losses and poor nitrogen recovery by plants. Under pasture, main losses of nitrogen are from volatilisation and leaching with minimal losses from denitrification and erosion.

Nitrogen losses may be increased or decreased depending on the climate and soil conditions. Weier (1994) found that small amounts of water in sugar cane trash resulted in over 30% loss of ammonia and losses increased to 44% in winter if soil was wet before fertiliser application. Decreased volatilisation resulted from heavy rain, which decreased losses to 17% and dry conditions where little ammonia was lost. Application of nitrogen fertiliser as ammonium sulphate rather than urea decreased volatilisation losses to < 2%.

Denitrification data are difficult to collect due to prohibitive costs. Losses reported in Weier's review (1994) ranged from 1-12 kg N/ha/week in the surface soil, which was equivalent to 2% to 20% of the nitrogen applied. Denitrification losses may also occur at depth in the soil and in some soils can be a substantial loss, e.g. Weier (1994) notes losses of 30 kg N/ha/yr have been reported in a Black Earth, which has pronounced shrinking during dry periods and swelling during wet periods leading to alternate aerobic and anaerobic conditions.

Denitrification may occur at depth in the soil where anaerobic conditions exist

Nitrogen movement through the soil is predominantly in the form of nitrate. Weier (1994) notes that low pH and low organic matter at depth leads to net positive charge and nitrate ions are electrostatically held thus reducing leaching out of the root zone. Some soil has significant anion exchange capacity, which can also reduce nitrogen losses through leaching. Retention of stubble and no till also reduce nitrogen leaching.

Stevens *et al.* (1999) conducted subcatchment scale trials at sites near Myponga and Mount Bold in South Australia's Adelaide Hills, to determine the proportion of nitrate, phosphorus and dissolved organic carbon (DOC) that moved from a duplex soil from runoff and throughflow and to characterise the soil properties that influenced their movement.

The two trial sites varied slightly with the Myponga site comprised of loamy sand/sandy loam with a sharp boundary to heavy clay. The Mount Bold site had silty clay loam topsoil less than 20 cm deep over well-structured medium heavy clay. Nitrate at both sites moved predominantly as throughflow in the B horizon with a lesser proportion moving at the interface between the A and B horizon. At the Mount Bold site, nitrate also moved as overland flow (runoff). These results show that nitrate is quickly moved down through the profile with the initial leaching front. However, where the topsoil is shallow and infiltration is restricted, some nitrate may be removed in overland flow. Ridley *et al.* (2001) found similar results with high nitrate losses measured as throughflow.

Nitrate may be lost as overland flow, though a greater proportion is usually lost in throughflow.

Nitrate concentrations in runoff were reported for a Victorian dairy site in a sub-catchment of the Murray-Darling Basin (Nexhip *et al.*, 1997). The average annual outfall from the subcatchment was 12 kg N/ha, with the runoff concentration varying from 3-17 mg N/L. The highest concentration in the runoff was measured where irrigation was undertaken immediately after fertiliser application. Following grazing of the pasture, the runoff concentration was between 3-6 mg/L. Nitrogen concentration in runoff as a result of winter rainfall ranged from 0.6-9 mg/L. From this study, it is evident that fertiliser application timing is critical to reducing nitrogen runoff losses.

On a catchment scale, Viney *et al.* (2000) measured and modelled two subcatchment inputs to the Swan and Avon Rivers in Western Australia. The large catchment contributed 62% of streamflow discharge and 54% of nitrogen inflow, while a small catchment with sandy soil contributed 6% of the freshwater flow and 10% of the nitrogen flow. From both catchments, the nitrogen input was predominantly particulate, i.e. > 0.45 µm in size and usually of organo-mineral form.

Absorption Trenches

Whelan and Barrow (1984a) studied the movement and transformation of nitrogen under seven septic tank installations on the sandy soil plains surrounding Perth. The nitrogen in the wastewater from the household was converted predominantly to ammonia through anaerobic digestion in the septic tank. Geary (2003) suggests 75% of the nitrogen is as ammonia with the remaining 25% as organic nitrogen, which are similar concentration to those quoted by the Virginia Division of Health (VDH, unknown) of 75-85% ammonia and 15-25% organic N.

In the absorption trenches, Whelan and Barrow (1984a) measured large quantities of organic nitrogen accumulated in the soil beneath the drain. The increase in organic N did not occur for more than 1 m depth. Below this the N was present almost entirely as mineral N.

From directly below the layer of slime in the base of the absorption trench, the sand transformed from fully saturated to 20% air-filled pores which was associated with a rapid increase in nitrate concentrations. Difference were found between the input end and output end of the trench with ammonia not converted to nitrate at input end, which was believed due to anaerobic conditions. Shortest distance reported for complete oxidation of ammonia to nitrate was 0.6 m through unsaturated soil. Oxidation of nitrogen resulted in a reduction of pH by up to 2 units.

The sandy soil provided little retention of inorganic N in the profile and hence nitrogen either moved to groundwater or was used by plants. Brown and Thomas (1978) reported grass grown over a septic tank drainage line took up 46% of N applied to soil. However, to exploit fully, the vegetative material would have to be removed regularly.

2.2.5 Plant Uptake

Plant uptake also reduces the nitrogen in the soil and hence limits the available nitrogen for leaching. The amount of nitrogen able to be taken up varies between plants. Ridley *et al*. (2001) found nitrate loss was less under perennial pasture than annual pasture and higher with the application of lime (lime increased soil pH ($CaCl_2$) from 4 to 5.5).

The nitrogen accumulated by plants varies between species. Phalaris can accumulate more nitrogen than cocksfoot or annual ryegrass, with significant differences found for most years (Ridley *et al.*, 1999). Phalaris generally had higher dry matter yields, though cocksfoot N concentrations in herbage were equal to or higher than phalaris and annual ryegrass. Annual ryegrass generally yielded less and accumulated less N than perennial pastures, except in spring when it was equivalent.

Plant species have different abilities to take up nitrogen.

A direct relationship between nitrogen available for plant growth and nitrogen lost in leaching does not always exist. Nitrogen fertiliser was applied to a grass pasture in New Zealand as a one-off application at a rate 2.5 times greater than was estimated to be fixed by clover in a mixed pasture (Ruz-Jerez *et al.*, 1995). The nitrogen lost from the grass pasture was 6-7 times greater than the nitrogen lost below the mixed pasture. The N concentration in the runoff from the grass pasture exceeded 10 mg/L, while the concentration in mixed pasture, where the N was released over the growing season by the clover, remained below this level.

Less nitrogen is lost due to leaching where the nitrogen is supplied incrementally compared to inorganic fertiliser applications.

2.2.6 Summary

- Nitrogen transformation in the soil is controlled by micro-organisms;
- Organic N is hydrolysed to ammonium when an energy source is available to the heterotrophic micro-organisms;
- Ammonium is readily oxidised to nitrate (via nitrite) in warm, moist, aerobic conditions by specialised bacteria;
- Ammonium may be held in the soil on the cation exchange sites, taken up by plants or volatilise, depending on concentrations and moisture conditions;
- Nitrate is readily leached from the soil, taken up by plants or denitrified. Denitrification may occur at depth in the soil where anaerobic conditions exist;
- Nitrate may be lost as overland flow, though a greater proportion is usually lost in throughflow;
- Plant species have different abilities to take up nitrogen;
- Less nitrogen is lost due to leaching where the nitrogen is supplied incrementally (as happens with fixation by clover) compared to inorganic fertiliser applications.

2.3 Phosphorus Cycle

The phosphorus (P) reactions in soil are more complex than nitrogen. P may be strongly adsorbed to clay particles and organic matter or may form compounds with cations in the soil. In the soil system, P exists almost entirely as the anion phosphate (PO_4^-). Sources of P are rainfall, plants, fertilisers, animal waste products and soil. Nash and Halliwell (2000) note atmospheric P input has been reported at 4 kg/ha/yr though this is believed to be excessive and overall rainfall is not believed to be a significant contributor. Plants, animal wastes and soil all contain appreciable concentrations of P. A simplified schematic of the P cycle is shown in Figure 2.3.

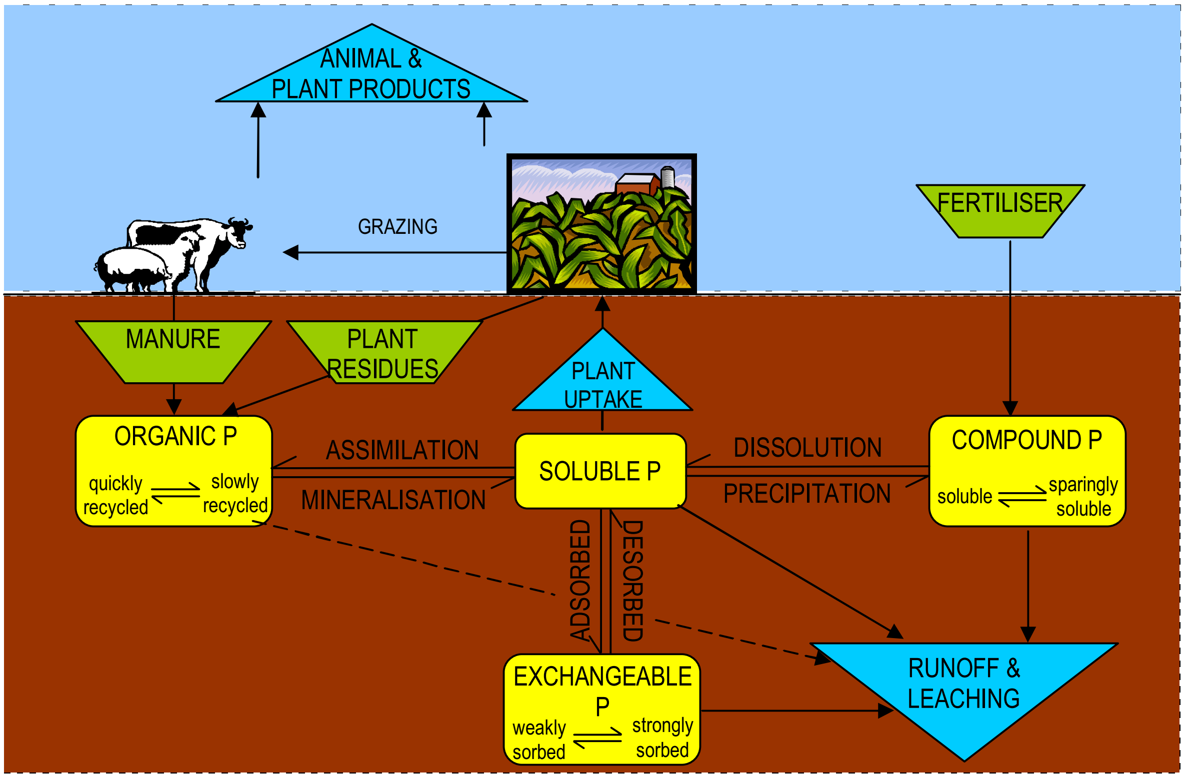

Figure 2.3 The Phosphorus Cycle, showing inputs to ( ), transformations in ( ) and losses from ( ) the soil.

Losses of P are by plant uptake and export of plant and animal products and by runoff and leaching beyond the root zone. There is no gaseous loss of P from the soil. Losses of P from the soil are often considered small as they are not agronomically important. However, these small concentrations may be environmentally important, as only small concentrations of dissolved P in surface water increase the probability of algal bloom. Heathwaite and Sharpley (1999) define critical P concentrations for prevention of eutrophication in water as 0.01-0.02 mg/L, which is an order magnitude lower than the critical soil P concentrations for plant growth of 0.2-0.3 mg/L.

2.3.1 Influences on Phosphorus Forms

The concentration of soluble P in the soil is affected by the concentration in solution, the quantity of P in the exchangeable, compound and organic P phases that is able to equilibrate and the P sorptivity or buffering capacity of the soil (Holford, 1997).

Phosphorus in the soil is predominantly present as organic P, which represents approximately 60% of the total P found in the soil (Nash and Halliwell, 2000). Sharpley and Halvorson (1994) note that moderately labile P contributed 83-93% of P mineralised and that mineralisation rates from 15-33 kg P/ha/yr have been reported in literature with higher rates for unfertilised soil than fertilised soil. Manure application has been found to increase the available P by 12-27 kg P/ha for every 100 kg P/ha added.

Perrott *et al.* (1992) found that organic P and organic matter increased in the soil in winter in New Zealand. They suggested that this was probably due to reduced microbial and enzymatic activity and higher numbers of bacteria compared to fungi, as bacteria contain more P than fungi. In the spring, the higher temperatures and increased plant root growth (which provides root exudates as a microbial substrate) leads to increased bacterial growth and activity which in turn promotes the mineralisation of organic P. Also protozoa grazing on bacteria would reduce microbial P and release inorganic P (i.e. soluble, exchangeable and compound forms). Event related maxima and minima were related to increases in bacterial activity and subsequent grazing by protozoa.

In addition to microbial effects, Weaver *et al.* (1988) suggest that the rate of removal versus the rate of conversion to soluble P is the reason for the low concentrations during winter in the sandy soil of the Peel Harvey Catchment. Fertiliser application at the beginning of winter increases the soluble, rapidly released and slowly released P forms. The onset of winter rains results in significant leaching which removes the soluble P at a faster rate than supplied by the rapidly and slowly released pools. A break in rainfall allows the soluble P to increase once again. By the end of winter, the total P in the soil is at a low level. Some increase of P occurs during summer from plant death but is only slowly converted to soluble P due to limited water availability affecting microbial decomposition.

The studies by Perrott *et al.* (1992) and Weaver *et al.* (1988) showed that climatic difference between years would have an affect on the magnitude of these events and that flux between the phases of P in the soil changes throughout the season as the soil attempts to maintain chemical equilibrium.

The concentration of exchangeable P in the soil is affected by the P sorption capacity, with field trial sites in Victoria showing that exchangeable P concentrations remained two to three times higher in soil with low to moderate sorption capacity than soil with high sorption capacity for 30 months after initial fertiliser application (Burkitt *et al.*, 2002). When combined with exchangeable hydrogen (H) or organic carbon, the decline in exchangeable P over the growing season was more accurately explained. More regular applications compared to a large single application resulted in higher exchangeable P concentrations which became more marked with time. The exchangeable P concentrations were not affected by the type of fertiliser applied, i.e. single super or triple super, or the application of 5 t/ha of lime.

P is more available in the soil when fertiliser is applied more frequently compared to one larger application.

Chemical and biological activity of the soil regulates the nature and extent of nutrient transformation. Power (1990) notes that as a result of drying, precipitation of salts can immobilise nutrients in forms that are not readily available to plants. Changes in water and aeration may result in oxidation-reduction reactions that convert some elements (such as Fe, Mn and other) from plant-available to unavailable forms. Soil drying can result in precipitation of calcium phosphate. In temperate regions, precipitated calcium phosphates are as octocalcium phosphates, which are slightly soluble. This mineral exists in equilibrium with soluble and adsorbed PO_4, with equilibrium concentrations varying with water content, salt concentration, pH and other factors. Much of P in this form can become available to plants. Under wetter conditions, Bramley and Barrow (1992) found greater P sorption compared with dry soil conditions.

P transformation in soil is dependent on a number of chemical and micro-biological factors.

Phosphorus Sorptivity

The ability of the soil to adsorb P is described in terms of its capacity (the amount that can be adsorbed) and its strength (the strength with which the P is held by the soil). P sorption primarily occurs by covalent bonding of phosphate anions to hydrous oxides of iron, aluminium and calcium (Moody and Bolland, 1999). One-quarter of the total number of adsorption sites in a soil will exhibit strong adsorption while the remainder will show weak adsorption, based on tests to define the soil's P sorption (Holford, 1997).

Holford *et al.* (1997) found that effluent application in the longer term reduced the soil's P sorption capacity and strength. On a texture contrast soil (i.e. lighter texture topsoil over clay subsoil) from a golf course at Bermagui, NSW, leaching occurred to 70 cm after 3 years effluent application even though only 17-38% of sorption capacity of each overlying horizon had been saturated. After 12 years of effluent application, the sorption capacity had been further reduced by approximately 38%, ranging from 74% reduction in capacity in the surface to 17% reduction at 70-100 cm. The soluble P did not increase in the 70-100 cm depth interval over this time, however only two-thirds of the P applied in effluent remained in the profile, suggesting leaching beyond 1m had occurred. Holford *et al.* (1997) suggested that the organic cations in the effluent may lower the P sorption capacity and strength by interacting with these sites, though this appeared to be partially balanced by the creation of new sorption sites by adding organic matter.

Leaching of P may occur prior to soil sorption capacity becoming saturated.

2.3.2 Runoff and Leaching

The standard view of P movement is that leaching only occurs in coarsely structured soil due to rapid infiltration of water and in sandy soil due to lack of P-sorption sites. P moves only a few centimetres in soil where there is capacity to hold P and clay minerals are present. Haygarth *et al.* (1998) found Olsen P concentrations were two times higher in the surface 1 cm of soil (12-18 mg/kg) than at 2 cm (7 mg/kg). P accumulation in the top few centimetres is due primarily to time-dependent adsorption and fixation processes occurring mainly at the surfaces of iron and aluminium oxides and calcium and magnesium carbonates. Soil with high organic content may have reduced P sorptivity, due to soluble organic anions coating active sites for P adsorption.

The concentration of P measured in the soil may not be directly related to the concentration measured in the runoff. P may be measured by various methods that extract varying proportions of P from the soluble, exchangeable, compound or organic phases. In increasing order of P extracted the methods are: Distilled water, Morgan, Olsen, Colwell, Mehlich III, Bray-Kurtz P1 and ammonium-oxalate methods (Pote *et al.*, 1999). Correlation between all methods and the P measured in runoff has been reported as poor (Stevens *et al.*, 1999; Pote *et al.*, 1999). The strongest correlation was found between water soluble P in soil and dissolved reactive P in runoff suggesting that the concentration in runoff is not contributed to by any exchangeable sources and no further reactions were occurring in the runoff water.

Soil P concentrations are not a good predictor of P concentration in runoff.

The concentration of P in runoff in Victoria has been measured varying from 3-21 mg P/L, with the highest concentrations measured immediately after fertiliser application (Nexhip *et al.*, 1997). The runoff following grazing contained P concentrations ranging from 2-3 mg/L and from winter rainfall varied from 0.2-5 mg/L. The P concentrations in the runoff were linearly related to stocking rate and the P concentration in winter runoff was related to soil P concentration as measured by the Olsen method. No apparent relationship was observed by Nexhip *et al.* (1997) between soil cover and P concentration in runoff, which supports the theory that erosion was not the major mechanism of transport.

Extensive trials on understanding the processes in nutrient mobility have been undertaken in the Adelaide Hills of South Australia (Kirkby *et al.*, 1997; Fleming and Cox, 1998; Stevens *et al.*, 1999; Cox *et al.*, 2000; Cox and Pitman, 2001; Fleming and Cox, 2001). The trials were all based on texture contrast or duplex soil, i.e. where sandy topsoil overlies clay subsoil. Phosphorus was measured as total P, particulate P (i.e. the P retained in a 45 µm filter), dissolved P (that which passes a 45 µm filter) and molybdate reactive P, which is inorganic P and believed to be indicative of the likelihood of the P increasing the probability of algal bloom.

At one trial site, the upper and mid slope soil were acidic and relatively well-drained while lower slopes were saline and neutral to alkaline and prone to water-logging. Kirkby *et al.* (1997) found in soil cores taken from the site that "new" water was measured in leachate after first day even though less than one pore volume of water was applied, suggesting that considerable bypass flow was occurring.

Comparison was also made between cores taken from the upper, mid and lower slope positions (Kirkby *et al*., 1997; Cox *et al*., 2000) and for cores from differing soil types (Cox *et al*., 2000). The time for leaching to commence was longer in cores taken from the lower slope due to increased thickness of the A and B horizons and when the soil was initially wet. An increased time till drainage leads to greater opportunity for the P to be adsorbed. The A horizon of the different slope positions did not show differing flow paths. However, the B horizons did have different flow suggesting increased residence time due to thicker horizon and a reduction in bypass flow in the cores from the lower slope position. Where the B horizon had strong macroporosity, this led to high P concentrations in the leachate regardless of whether the soil had high P sorption capacity, showing that P sorption capacity is not a good indicator of the amount of P leaching for soil with strong preferential flow paths (Cox *et al*., 2000).

Water flow may bypass the soil mass in macropores leading to increased P concentrations in the leachate, regardless of the soil's P sorption capacity.

Intermittent watering of the soil cores generally resulted in less P leached through the cores compared to continuous watering. In sand soil types, Cox *et al.* (2000) found that prewetting the core prior to fertiliser application reduced P in leachate from 61 mg/L when dry to 16 mg/L, with 46% of initial water from soil storage. In gradational soil types (i.e. with gradually increasing clay with depth), high P adsorption capacity and low macropores allowed little P to leach (1 mg/L in initial flush) and had high residence time resulting in good P sorption. A loamy soil type with high P sorption capacity and strong macroporosity had an initial flush of P in leachate (20 mg/L) but then reduced with time indicating that the transport pathways were through the macropores. Prewetting led to a dramatic reduction in leached P. The breakthrough of new water occurred more quickly in dry cores than wet suggesting that dry flow is dominated by macropore flow and in wet cores it is dominated by matrix flow. Haygarth *et al.* (1998) found similar results that suggested two mechanisms for P movement: large discharge and fast transfer when storms occur shortly after P application and base flow resulting from longer term soil dynamics.

Intermittent irrigation and application of P to moist soil decreasing the amount of P leached.

The amount of P recovered in leachate through the A horizon trended down slope with highest recovery in upper slope and lowest in footslope with intermediate values in between. This followed the depth of the A horizon with shorter residence time leading to less opportunity for P adsorption. P recovered from A+B cores was markedly higher in upper slope cores than mid and lower slope and control. Slightly more P was recovered from foot slope than higher positions suggesting higher native P in the lower landscape position as a result of translocation. Thinner A and B horizon (as a result of pedological processes) and higher permeability of B horizon in upper slope position compared to mid, lower and foot slopes decreased residence time of irrigation water which reduced time for P to adsorb onto soil. A linear decrease in P concentration indicated that the pores are able to support P transport for a considerable time (Cox *et al.*, 2000).

Fleming and Cox (1998), Cox and Pitman (2001) and Fleming and Cox (2001) compared the proportion of P in overland flow and through flow from a pastured catchment. Through flow was the dominant flow path for water and chemical species, reflecting the duplex texture of the soil. In low to average rainfall years, through flow is mainly responsible for transporting chemicals. The pH of overland flow reflected the pH of the topsoil, while the pH of the subsoil was variable. In dry years the throughflow was alkaline reflecting the pH of the subsoil. In wet years, the pH was acidic to neutral, reflecting the leaching of humic substances. The P was almost all as overland flow and did not vary with pasture species. Losses of P were up to 2.4 kg/ha/yr. Fleming and Cox (1998) also measured a number of other cations and anions in the overland flow and through flow and found that highly mobile elements, such as Na, K, Ca and Mg, had higher concentrations in throughflow than overland flow.

Stevens *et al.* (1999) compared two texture contrast soil types at Myponga and Mt Bold. The Myponga site had lighter textured and thicker A horizon over a heavy clay B horizon while the Mount Bold site had siltier, thinner A horizon over a well-structured B horizon. The dissolved P in overland flow and through flow at the Myponga site was almost 0 kg/ha compared to Mount Bold where the majority of dissolved P was removed as overland flow, with total P losses of 0.07-0.17 kg/ha. Differences were evident between the two subcatchments at the Mount Bold site, which we reattributed to variation in subcatchment shape or variation in soil P concentrations. The east subcatchment (which had lower concentration) was longer than west and has deeper topsoil layer and so would have longer residence time for P to be adsorbed.

The proportion of dissolved P compared to total P was higher at Mt Bold at 50% dissolved compared to Myponga, where 8% of total P was dissolved. The authors noted this result as unexpected as Myponga A horizon was sandier, which should result in higher proportion of dissolved, and Mt Bold has hard-hoofed animals grazing which should have result in a higher proportion of particulate P. However, average turbidity of samples from Myponga was higher than Mt Bold which may be due to dispersion of sodic B horizon.

Fleming and Cox (2001) noted that the duration of saturation of soil between years affected proportion of P as dissolved or particulate P and in their trial 45% of P lost was dissolved. The dissolved P concentrations measured in runoff were increased with increasing slope and volume of runoff. Particulate P concentrations were increased with the peak volume of the runoff event and where peak flow rate was a small proportion of the total volume of event and the event was short, i.e. P concentrations decreased with increasing time the rainfall event occurred. Nash and Murdoch (1997) found the variation in P concentrations in larger volume runoff events was associated with flow rate (negative relationship), cumulative flow (neither positive nor negative suggesting a constant supply of P was available) and time of runoff commencement from commencement of the storm (positive relationship). Time of runoff was best predictor for P concentration in runoff.

The trials in the Adelaide Hills all showed that increased residence time of the leachate in the soil increases the potential for P sorption and decreases the P in leachate. Increased residence time may be due to soil depth, texture and structure. Also the P adsorption is related to clay content, density and exchangeable sodium percentage.

Soil hydrology and not chemistry is the best predictor of P leaching.

Kirkby *et al.* (1997) applied P in solution and as a powder and compared to unfertilised soil cores. The P leached from cores of the A horizon was similar for control (no P fertiliser added) and P added as solution, with 5 mg leached after 25 days. Where the P fertiliser was added as a powder, an initial flush of P occurred for 5 days and resulted in 17 mg P leached after 25 days. P leached from the cores of the A and B horizon was similar between control and treated cores.

Fertiliser in solution is more readily sorbed than as a powder and hence less P may be leached from the soil.

The form of the P in the leachate varied between the fertiliser treatments. In the control core, 62% was dissolved and remainder particulate P while in fertiliser cores, P was predominantly dissolved. In the A horizon at all topographic positions, the dissolved P was mainly molybdate reactive, suggesting that it was derived from the fertiliser applied. In the combined A and B horizons, only the core form the upper slope had molybdate reactive P concentrations in excess of control, further suggesting that the P leached from the mid, lower and foot slope was native P and not applied in fertiliser. The concentration of iron and aluminium was also measured in the leachate, as it is indicative of clay particle movement. These tests showed that the phosphorus is not moving in association with soil particles.

In contrast to the Adelaide Hills, the total phosphorus concentration in runoff measured from a site in Victoria varied from 2.5 mg/L to 7.4 mg/L, of which 93% was dissolved and 89% was dissolved and reactive, which indicated P would be easily taken up by aquatic plants (Nash and Murdoch, 1997). At a site in the UK, dissolved P represented 69% of total P in surface pathways (Haygarth *et al.*, 1998). Of the total P lost from the site, 33 to 50% of P was as organic P and not only inorganic P.

The amount of P dissolved or attached to particles in runoff or leachate varies between sites.

The average phosphorus lost per unit area reduces with increasing catchment size as the heterogeneous land use conditions often mask the effect of individual land uses. The average annual P lost from a subcatchment of the Murray River in Victoria was 7 kg P/ha (Nexhip *et al.*, 1997). However, the P concentration in surrounding streams was generally lower due to dilution but also possibly due to dissolved reactive P sorbing to sediment in streams and becoming particulate P (Nash and Murdoch, 1997). Export from a pastured site in the UK was estimated to be 3 kg P/ha/yr from an undrained site and was reduced in a drained site (i.e. one with artificial subsurface drains) reflecting greater opportunity for P sorption (Haygarth *et al.*, 1998). In the Swan River in Western Australia, a small catchment with sandy, low P sorption soil combined with horticulture and increasing residential use contributed only 6% of the estuaries fresh water but 36% of its total P and 62% of the soluble phosphorus flow (Viney *et al.*, 2000). In comparison, a large catchment of the Avon River in Western Australia contributed 62% of streamflow discharge and contributed 28% of P inflow, 75% of which was particulate (Viney *et al.*, 2000).

Monitoring of 42 subcatchments of the Peel-Harvey estuary in Western Australia was undertaken by Summers *et al.* (1999). A range of parameters of each catchment were measured, including area, soil type, phosphorus sorption, waterlogging, pasture, though none accounted for a large proportion of the variation in total P in the streams. Sandy soil with low P sorption had the highest concentration of P in runoff but the lowest total volume of runoff. The area closest to the drainage system had the most influence on the concentration of P in the stream with factors such as absence of native vegetation, presence of pasture and sandy soil associated with increased P concentration in the stream. Scale effects, variation between sites, differences between years, hydrologic idiosyncrasies, errors in measurement and differences in management and history can produce large errors resulting in inconsistent results. Comparison of management solutions is best on the same scale and preferably on a small scale.

P concentration in streams is affected by the land uses in the catchment, with the area closest to the stream having the greatest effect. However, the concentrations at the stream can not be directly linked to the concentration at the boundary of each land use.

Absorption Trenches

The source of P in grey water is predominantly from detergents, which may contain 19% P as pentasodium phosphate. In a study of septic tanks in Perth, the concentration in grey water varied from 8-18 mg/L compared to black water which contained 24-29 mg/L and from combined systems, which contained 15-23 mg/L (Whelan and Barrow, 1984b). The total P in the effluent is predominantly ortho-P (85%) with the remaining 15% as polyphosphates or organic P (VDH, unknown).

Whelan and Barrows (1984b) study of absorption trenches found that in systems with a long history of septic tank effluent disposal, the P concentration in soil solution was similar to concentration in effluent. In systems with a short history of disposal, the P concentration in the soil decreased over a short depth (1 cm). Increases in organic P were confined to a layer approximately 15 cm thick directly below the absorption trench. The inorganic soil P varied down the profile with concentrations reflecting the P sorption capacity of the soil and not the soluble P content, which was the same at various depths. P compounds, from precipitation of iron and calcium phosphates may have occurred in some cases in the slime layer. In the systems the P did not move laterally due to water movement through the sand being predominantly vertical. Phosphorus was found to have leached to groundwater within a few years of the system being installed, as shown by a similar concentration in effluent and soil solution, soil concentrations matching the sorption characteristic and the total P stored at the site was less than the amount applied.

The phosphorus movement from septic effluent is affected by reactive Al, Fe and Ca concentrations, flow rate, pH, soil particle size distribution and depth to groundwater. P is sorbed by reactive Al, Fe and Ca, which decreases its availability for leaching. Increased flow rate through the soil due to soil texture, soil structure or presence of macropores or high water tables increases the potential for P movement. Lateral movement of P has been observed in some studies at horizontal distances greater than 30 m (VDH, unknown).

2.3.3 Plant Uptake

The uptake of P by plants is decreased as the P sorption strength of the soil is increased (Holford, 1997). However, as the root system in a ryegrass trial became more extensive, the uptake became less reliant on sorption capacity as the roots were able to access an increasing proportion of the total P reservoir. Plant uptake of P is reduced by low temperature, high compaction, high and low soil pH, high clay content and high iron, aluminium and calcium carbonate content and increased in the presence of ammonium due to the creation of an acidic environment around the root after ammonium uptake (Sharpley and Halvorson, 1994) and the plant species grown.

Plant uptake of P is dependent on the plant species and the availability of P in the soil.

The effect of withholding P fertiliser from pastures in New Zealand was measured by Perrott *et al.* (1992). Plant yield and P uptake were higher from fertilised plots compared to unfertilised plots. The P was taken up from the inorganic P resources and/or fertiliser but not from the organic P, which accumulated over the 2 years of the trial.

Plants use inorganic P from the soil solution.

Plants may also contribute P to runoff, with concentrations reported between 2-150 mg/L (Nash and Halliwell, 2000) as P may accumulate on the leaves of the plant. Plants grown in soil with high concentration of available P had more P with more held in water soluble form though lost the same percentage P as plants grown on lower P soil. P lost from plants in runoff was increased after mowing.

Sharpley (1981) found that the soluble P in leachate varied little between soil types but varied depending on plant species. The amount of soluble P in the runoff increased with plant age and soil water stress. In 30 minutes of simulated rainfall, the P concentration in leachate dropped markedly from 0.04-0.11 mg/L during first 5 minutes to 0.02 to 0.04 mg/L (depending on plant species) and had almost reached a steady concentration at the end of 30 minute rainfall event. The mean concentration of P in leachate remained constant where rainfall events were over 24 hr apart, while rainfall events at 0.5 hr, 2 hr and 6 hourly intervals showed decreasing trend for P concentration in leachate. This suggests that P is secreted onto the leaves slowly and takes 24 hours to return to pre-rainfall amounts.

Plants contribute P to runoff.

2.3.4 Summary

- P always exists as phosphate minerals in nature;
- Solubility varies and P tends to transform from sparingly soluble to insoluble with time;
- P transformation in soil is dependent on a number of chemical and micro-biological factors.
- Important soil properties that affect P solubility are pH, concentration of iron (Fe), aluminium (Al) and calcium (Ca) and the nature and surface area of soil particles. Some P is adsorbed to the surfaces of clay minerals, mainly Fe and Al hydrous oxides and organic matter complexes. In acidic soil, Al and Fe phosphates form and in alkaline soil Ca and magnesium (Mg) phosphates are more likely to form and be adsorbed on the surfaces of carbonates;
- Organic P is a major and very stable P component of most soil, especially in acidic pH and OM and N content is high;

- Leaching of P may occur prior to soil sorption capacity becoming saturated.
- P is more available in the soil when fertiliser is applied more frequently compared to one larger application.
- P is mostly lost through runoff. In pastures, P is predominantly dissolved while in disturbed soils, eg. ploughed fields, the P is predominantly particulate;
- Soil P concentrations are not a good predictor of P concentration in runoff;
- Water flow may bypass the soil mass in macropores leading to increased P concentrations in the leachate, regardless of the soil's P sorption capacity.
- Intermittent irrigation and application of P to moist soil decreasing the amount of P leached.
- Soil hydrology controls the movement of P in the soil, especially for soil with strong macroporosity. Large discharge over a short period of time occurs due to preferential flow paths. In the longer term, once the soil is moist, the flow is predominantly through the matrix allowing greater time for P sorption to occur;
- Fertiliser in solution is more readily sorbed than as a powder and hence less P may be leached from the soil.
- The amount of P dissolved or attached to particles in runoff or leachate varies between sites.
- P concentration in streams is affected by the land uses in the catchment, with the area closest to the stream having the greatest effect. However, the concentrations at the stream can not be directly linked to the concentration at the boundary of each land use.
- Plant uptake of P is dependent on the plant species and the availability of P in the soil.
- Plants use inorganic P from the soil solution and can contribute P to runoff from their leaves.

3. Nutrient Management Strategies for Effluent Irrigated Sites

3.1 Effluent Treatment Systems

Failure rate for septic systems has been associated with detrimental impacts on human health but can also result in excessive nutrients being applied to the disposal/irrigation area. A review of on-site systems in Sydney's drinking water catchments has shown that poor design, operation and maintenance are the major causes of water pollution from these systems (Charles *et al.*, 2001). A survey of 48 septic tanks, including sampling of the effluent, found high variability between tanks, with all chemical constituents showing log normal distributions (Charles *et al.*, 2003). For example, the concentration of suspended solids varied from 2 mg/L to 29,000 mg/L, with a geometric mean of 177 mg/L. NSW now operates a program, known as SepticSafe to assess on-site systems, manage cumulative pollution impacts, implement sustainable on-site sewage management practices and assess the impact on rivers (Geary, 2003).

The concentration of nitrogen and phosphorus in effluent varies for different treatment systems. Gardner *et al.* (1997) compiled data from various sources and presented a comparison of raw effluent quality with effluent from septic tanks, aerated wastewater treatment systems (AWTS) and sand mound, which are presented in Table 3.1. Nitrogen concentrations are not markedly different between the treatment options. However, the nitrogen in septic tank effluent is predominantly ammonium, while AWTS and sand mound effluents are 80-85% nitrate. As a result, application of effluent from AWTS and sand mounds provides nitrogen in a form that is readily available for plant uptake and leaching. Phosphorus concentrations are slightly different between treatment options, though 85-90% of phosphorus is present as phosphate for all treatment systems.

Table 3.1 Comparison of effluent quality from different on-site treatment systems.

Parameter	Concentration (mg/L) in effluent			
	Raw Effluent	Septic Tank Effluent	AWTS [a] Effluent	Sand Mound Effluent
Biological Oxygen Demand	300-340	120-150	5-80	1-10
Suspended Solids	260-300	40-190	5-100	5-20
Total N	50-60	40-50	25-50	30-50
Total P	10-15	10-15	7-12	5-10

From Gardner *et al.* (1997)

a AWTS – aerated wastewater treatment system

The concentrations in effluent from an AWTS are more variable than septic due to potential for volatilisation, denitrification and precipitation and settling into the sludge fraction during "rest" periods. Tertiary treatment is required to reduce N and P concentrations in effluent, as secondary treatment is designed to reduce bacteria, BOD and suspended solids.

Various authors have shown that the concentration of septic systems is important in limiting potential environmental impacts from nutrient leaching. This research has suggested that septic tank and trench systems of 15 systems/km^2 or more are likely to be causing nitrate and bacterial contamination of local groundwater systems (DLP *et al.*, 1998). A study in Victoria by Rural Water Corporation (1993), as referenced by Gardner *et al.* (1997), found that where septic tank density was greater than 15 systems/km^2, the nitrate concentration in groundwater 10 metres below were up to 17 mg/L, which exceeds the ANZECC (1992) guidelines concentration for raw water for drinking purposes of 10 mg/L.

The system densities can be converted to minimum allotment sizes for various areas. The West Australian Water Authority has set a limit of 25 systems/km^2 where there are significant potable water supplies from groundwater, which is equivalent to allotment sizes of 4 ha. Where the land value exceeds the need for protection of groundwater quality, Gardner *et al.* (1997) suggest that the system density could be increased to 100 tank/km^2 , which is equivalent to 1 ha allotments, or where groundwater contamination is not an issue, 100-250 systems/km^2, equivalent to 4000 m^2 to 10000 m^2 may be permissible.

The system density may also be reduced by changing to a different treatment system. Gardner *et al.* (1997) suggest that where individual household wastewater is treated by transpiration beds or AWTS, where a density of 250-330 systems/km^2, equivalent to an allotment size of 3000-5000 m^2 may be environmentally sustainable. However, many authors note a higher failure rate with AWTS for individual households due to poor operation and management by the householder compared to septic tanks and trenches.

3.2 Site Selection

Site selection has an important role in minimising the potential for nutrients to move through the soil. The research trials undertaken in the Adelaide Hills showed that soil texture, soil structure and depth affect the movement of water through the soil and varies with position in the landscape, hence the application of effluent should be matched with the most appropriate topographic and soil features.

The ideal soil would have a thick loam to clay loam A horizon, overlying a weakly to moderately structured clay B horizon, with no or few macropores. The A horizon should have good permeability to allow rainfall to infiltrate into the soil, reducing runoff, but sufficient clay to sorb P and hold cations, such as ammonium. The subsoil should have very high P sorption and ensure that flow is predominantly matric flow and short circuiting can not occur.

3.2.1 Site Limitations Criteria for Effluent Application

Hird *et al.* (1996) have compiled site selection criteria for sites to be irrigated with effluent. These criteria include soil and landscape factors to maximise the utilisation of nutrients in the effluent and minimise the potential for nutrients to runoff to surface water or leach to groundwater and are shown in Table 3.2. The criteria are based on degree of limitation of various features. These criteria have been reproduced, with some minor changes, in the South Australian 'A Manual for Spreading Nutrient-Rich Wastes on Agricultural Land' (SA Govt, 2002).

Land with severe limitations should be excluded from effluent irrigation. However, management measures may be available to reduce the severe risk to a lower risk. For example, a steeply sloping site may be terraced reducing the risk to slight. Surface outcrop may only exist on a small portion of the land and hence only this area need be excluded and not the whole site. Effluent with a high sodium absorption ratio (SAR) may be suitable to apply to soil with a higher SAR without detrimentally effecting soil structure or increasing the SAR of the soil. Effluent with a low electrical conductivity (EC) may be used on soil with a higher EC, depending on the form of salts in the effluent, as this may reduce the EC of the soil, making it more suitable for plant growth.

Land with moderate limitation may be irrigated but it is likely that additional management practices will be required and greater justification for proceeding in this area should be given. For example, irrigation on areas with moderate to frequent flooding may be justified if the effluent is replacing inorganic fertilisers (resulting in decreased potential for leaching of nutrients), is of a quality that could be discharged to the river or is the only economically viable alternative and could be used in preference to directly discharging to the river. Management strategies to limit potential impacts to the adjacent river could include limiting irrigation to drier months, growing nutrient "hungry" plants, harvesting and removing plants and/or constructing sediment and erosion control structures.

Sites with slight limitation are most likely to maximise plant growth while minimising the need for special management practices.

This limitations table was based on trials in New South Wales, though as it is based on fairly generic soil properties it is likely to be applicable to South Australian soil also. However, this system does not take into account the current nutrient status of the soil or the presence of macropores. In addition, this table was compiled assuming the effluent was secondary-treated effluent from a wastewater treatment plant and hence the importance of various topographic or soil limitations would vary with the quality of the effluent produced. For example, effluent that meets fresh water criteria of the Environment Protection (Water Quality Policy) Policy (SA Govt, 2003) should not be limited by these topographic or soil features.

Table 3.2 Limitations for effluent application based on topographic and soil limitations.

Property	Limitation			
	Slight	Moderate	Severe	Comment
Topographic Limitations				
Slope % (Sprinklers)	< 6	6-12	> 12	
Flooding	None-rare	Occasional – frequent		Flooding
Landform	Hillcrests, convex slopes and plains	Concave sideslopes and footslopes	Drainage plains and incised channels	Erosion and seasonal waterlogging risk
Surface rock outcrop	Nil	0-5%	> 5%	Interferes with cultivation, risk of runoff
Soil Limitations				
ESP	0-5	5-10	> 10 (> 40 cm)	Structural degradation
EC (dS/m)	< 4	4-8	> 8	Excess salt restricts plant growth
Depth to high water table (cm)	> 90	45-90	< 45	Wetness, risk to groundwater
Depth to bedrock or hardpan (cm)	> 90	45- 90	< 45	Restricts plant growth
Hydraulic conductivity (mm/hr, 0-100 cm)	20-80	2-20	< 5	Excess runoff, waterlogging, acts as a poor filter
Available water capacity (mm/m)	> 200	< 200		Little plant available water
Bulk density (g/cm^3) sandy loam loam and clay loam clay	 < 1.4 < 1.6 < 1.8	 > 1.4 > 1.6 > 1.8		Restricts root growth
Surface Soil pH ($CaCl_2$)	> 6.0	< 6.0		Reduces optimum plant growth
CEC (cmol(+)/kg) for 0-40 cm	> 15	< 15		Unable to “hold” plant nutrients
P sorption index	> 4	< 4		Unable to fix phosphorus

From Hird *et al.* (1996)

3.2.2 Critical Source Areas for P movement

For P to have an environmental impact it is necessary to have a source of P and a mechanism by which to transport the P to surface or groundwater. Heathwaite and Sharpley (1999) define P source factors that describe catchment areas that have high potential to contribute P and P transport factors that determine whether potential is translated into loss. Coincidence of sources and transport factors are critical source areas (CSA). Identification of CSAs will enable resources for reducing P loadings to be targeted to areas where the most benefit will occur.

Heathwaite and Sharpley (1999) list the factors that influence P sources and transport as:

P sources	P transport
soil P concentration;	topography;
soil texture, structure and permeability;	incidence of surface runoff;
soil erodibility;	contribution from subsurface flow;
land use;	location relative to drainage network;
cultivation practice;	storm return period;
fertiliser or manure inputs, forms and timing;	precipitation duration, intensity and magnitude.
livestock grazing density.	

Sharpley and Halverson (1994) and Sharpley (1995) present a P index incorporating the factors affecting P sources and transport, which is being developed by Phosphorus Indexing Core Team (PICT, led by the United States Department of Agriculture Soil Conservation Service, National Water Quality Technology Staff). This index is presented in Table 3.3.

The indexing system was found to be closely related to actual P losses measured over 16 years. Sharpley (1995) notes that it is limited by its use of annual data, which does not take account of individual storm events or frequency and timing of fertiliser application, and by not accounting for dissolved P.

The site vulnerability determined from the combined ratings (Table 3.4), may be used to determine appropriate management options and to target management practices where most benefit would result. For example as the vulnerability increases from low to very high, the frequency of soil testing should increase. Soil conservation measures may be required for sites where the vulnerability is medium or greater. These measures may include reducing tillage, maintaining buffer strips, promoting grassed waterways and using high demand P crops.

This index is based on US data and is intended as a field-use index. This index may not be directly applicable to Australia and so an Australian-based system is currently being developed by the “Better Fertiliser Decision Working Group” through the Department of Primary Industries at Elinbank, Victoria.

Table 3.3 P index to rate the potential for P loss in runoff.

Site characteristic (weight)	Phosphorus Loss Potential (Value)				
	None (0)	**Low (1)**	**Medium (2)**	**High (4)**	**Very high (8)**
Transport Factors					
Soil erosion [a] (1.5)	Negligible	<10	10-20	20-30	> 30
Irrigation erosion [b](1.5)	Negligible	tailwater recovery for QS < 6 for very erodible soils or QS < 10 for other	QS > 10 for erodible soils	QS > 6 for very erodible soils	
Runoff [c] (0.5)	<0.1	0.1-1.0	1-5	5-10	>10
Phosphorus source factors					
Soil P test [d] (1.0)	<10	10-20	20-40	40-65	>65
P fertiliser application rate [e] (0.75)	None applied	<10	10-20	20-40	>40
P fertiliser application method (0.5)	None applied	Placed with planter deeper than 5 cm	Incorporated immediately before crop	Incorporated > 3 months before crop or surface applied < 3 months before crop	Surface applied > 3 months before crop
Organic P source application rate [e] (0.5)	None applied	1-15	16-30	30-45	> 45
Organic P source application method (1.0)	None	Injected deeper than 5 cm	Incorporated immediately before crop	Incorporated > 3 months before crop or surface applied < 3 months before crop	Surface applied > 3 months before crop

From Sharpley and Halvorson (1994); Sharpley (1995)

a Soil erosion (t/ha);
b Q is flow rate (gal/min) and S is furrow slope (%);
c runoff (cm);
d as measured by Mehlich III method;
e application rate (kg P/ha).

The total index rating for P losses in runoff is calculated as:

Total Index Rating = $\sum$ (weight × value)

Interpretation of the total index rating to determine the site vulnerability is defined in Table 3.4.

Table 3.4 Site vulnerability for P loss in runoff.

Site Vulnerability	Total Index Rating
Low	< 10
Medium	10-18
High	19-36
Very high	>36

3.3 Disposal Area Management

Management of the disposal area can aid in reducing the potential for nutrient movement to occur. Management options include:

- Limit nutrient loadings;
- Choice of irrigation system to facilitate N gaseous loss and limit leaching;
- Harvest plant and animal products to remove accumulated N and P in the plant matter.

Nutrient Loadings

Hird *et al*. (1996) present rates of nitrogen and phosphorus removal from typical effluent irrigation enterprises, as shown in Table 3.5. Nitrogen uptake varies from 0-630 kg/ha depending on the plant species grown and the proportion of the plant removed from the site. Phosphorus uptake is markedly less, varying from 0-62 kg/ha.

Table 3.5 N and P removal from effluent irrigation systems.

Enterprise	Yield (t DM/ha)	Total potential N export (kg/ha)	Total potential P export (kg/ha)	Comments
Maize	10	160	30	Grain only
Wheat	7	154	28	Grain only
Lucerne	18	630	56	
Mixed pasture (dairy)	24	568	62	36% of pasture recycled
Mixed pasture (beef)	24	307	34	60% of pasture recycled
Pinus radiata	10	20	3	Needles and branches
Turfed recreation areas	8	0-187	0-27	Yield is amount cut and export is dependent on removal of clippings

From Hird *et al.* (1996)

DM – dry matter

Gardner *et al.* (1997) calculated a loading rate from trenches of 2600 kg N/ha/yr and 420 kg P/ha/yr and for irrigation system from AWTS as 650 kg N/ha/yr and 105 kg P/ha/yr. These quantities are in excess of plant nutrient requirements, suggesting a large amount of N and P may be available to be leached from the soil. However, these loadings do not account for the availability of applied N, gaseous loss of N or sorption of P. NSW EPA (1995) suggests gaseous loss of N from secondary treated effluent may range from 15-25% and that 30-40% of the organic N applied will be mineralised in the first year. This would result in effective N loadings of 1700-1800 kg N/ha from septic effluent and < 550 kg/ha/yr for AWTS effluent. These are still greater than the estimated N export from a turfed area.

The nutrient removal of a grass buffer after the application of pig effluent at two rates was reported by Hubbard *et al.* (2003). The effluent was applied at two rates which were equivalent to 800 kg N/ha/yr, 215 kg P/ha/yr and 1030 kg K/ha/yr and double these amounts. The biomass of the grass increased with the increased application rate but the percentage of N and P in the biomass was the same for both treatments. This suggests that where N and P are sufficient in the soil, N and P removal is predominantly through increasing biomass for the grass species used. Other plants, such as lucerne have been noted as “luxury” accumulators of nutrients.

Irrigation System

The choice of irrigation system usually depends on the quality of the effluent to be disposed. However, some opportunities exist for choosing the proposed irrigation system. Subsurface application methods, such as trenches and drippers, tend to create saturated areas with high organic carbon content that favour denitrification losses (VDH, unknown) while surface application, particularly spray irrigation favours volatilisation. Kruger *et al.* (1995) estimated volatilisation losses from sprinkler irrigation of 15-40% compared to 1-5% for subsurface application while USEPA (2000) suggests 10% loss through ammonia volatilisation if the soil pH is high otherwise less to none if pH is acidic for treated effluent from on-site wastewater systems which is spray irrigated.

Intermittent watering was found to reduce P leaching through cores (Cox *et al.*, 2000) and wetting and drying cycles help to promote denitrification. As a result, continuous application of effluent should be avoided. More frequent applications promote denitrification and hence reduce the N available for leaching and uptake and allow greater residence time for P sorption to occur. USEPA (2000) suggests 25% nitrogen losses through denitrification for treated effluent from on-site wastewater systems which is spray irrigated.

Application of organic rich effluent has been found to result in clogging of pores (VDH, unknown). Removal of clogging would require periods of “rest” with various periods from 1 day to 6 months recommended in the literature. However, it should be noted that the clogging layer is required to filter organic matter and micro-organisms and release effluent as unsaturated flow in to the soil matrix below soil adsorption trenches.

3.4 Erosion Control

The soil lost from erosion at a particular site may be altered by changing the crop management or land management factor of the universal soil loss equation. The rainfall erosivity, soil erodibility and slope steepness and length are fixed for a given location, though land and crop management may influence these factors.

3.4.1 Crop Management

Vegetation can reduce erosion through:

- creating a canopy which reduces rainfall erosivity and hence detachment of particles through raindrop impact. A rapid increase in soil loss occurs where the cover declines below 75% (Cumming and Elliott, 1991);
- increasing surface “roughness” which reduces the velocity and erosivity of overland flow;
- binding soil particles together through root growth and provision of organic matter.

The type and management of the crop will impact on the potential for runoff to occur. Sediment lost in runoff from native grass pastures was 0.1 t/ha/yr compared to cropping which varied from 0.3 to 16.2 t/ha/yr depending on crop type and tillage (Sharpley and Halvorson, 1994).

Crops require sunlight, water and nutrients to grow at their maximum potential. The amount of sunlight supplied in a particular location is fixed. Water is provided by the rainfall and supplemented by irrigation to maximise plant growth. Plants require macronutrients (hydrogen, oxygen, carbon, nitrogen, phosphorus and potassium) secondary nutrients (sulphur, magnesium and calcium) and micronutrients (boron, chloride, copper, iron, manganese, molybdenum and zinc) for satisfactory growth (Sposito, 1989). With the exception of hydrogen, carbon and oxygen, all of these may need to be added to the soil as supplements to ensure maximum plant growth. The use of effluent for irrigation does not guarantee supply of all the nutrients required for plant growth in the amounts the plant requires nor at the time the plant can use them.

3.4.2 Land Management

Cultivation practices influence the amount of sediment entrained in runoff. In Daly Basin in NT Australia, conventionally tilled catchments produced 1.5-2 times more runoff and lost 1.5-6 times more soil than their no-till counterparts (Dilshad *et al.*, 1996). On cultivated fields, 60-90% of P loss occurs as surface runoff through attachment of to eroded soil particles, while for grasslands, woodlands and non-erosive soil, dissolved P loss in runoff is more significant (Heathwaite and Sharpley, 1999). The concentration of phosphorus lost from sites in the US decreased with decreasing cultivation, such that wheat grown with conventional tillage methods lost the most phosphorus in runoff followed by wheat grown with no-till methods and then native grass and "set aside" areas (i.e. undisturbed buffer areas) (Sharpley, 1995).

Smith *et al.* (1996) note that old-age systems which are no longer accumulating biomass or organic matter tend to have less ability to hold nutrients than intermediate systems which are still accumulating organic matter. They suggest the following progression occurs in natural systems:

1) old-age or steady state allow nutrients through with little net gain or loss;
2) disturbance of these systems leads to rapid loss of dissolved and particulate nutrients;
3) as system recovers, they accumulate nutrients;
4) return to steady-state conditions.

This suggests that disturbance of systems should be undertaken with great care but that allowing the system to partially recover would have benefits for nutrient retention. Trials undertaken on 18 sites in the US with significant P accumulation in the surface soil from manure application were mixed with lower soil, by ploughing to reduce P concentrations in runoff (Sharpley, 2003). In short term, ploughing increased total P in overland flow, however after 10 weeks the P concentration in overland flow reduced from a pre-ploughing concentration of 3.4 mg/L to 1.8 mg/L. Dissolved P concentrations decreased quickly after ploughing from a pre-ploughing concentration of 2.9 mg/L to 0.3 mg/L after one week. Particulate P concentrations increased after ploughing and only returned to pre-plough levels after over 40 weeks. Sharpley (2003) concluded that ploughing, as a one-off measure, can be used to minimise P losses in overland flow in P-stratified soil, when combined with adequate erosion control measure to limit sediment entrainment and movement of particulate P into surrounding waterways. Continued ploughing could result in excessive accumulation for the entire plough depth.

The construction of soil conservation banks reduces the effective slope length. Sediment losses have been reduced by ten-fold in catchments in the semi-arid tropics (Dilshad *et al.*, 1996). Soil conservation banks may also be used to direct water to areas of greater sorptivity or to sediment detention structures which can achieve > 70% reduction in nutrient and sediment, if appropriately sized (Weaver and Prout, 1993).

3.4.3 Runoff Concentration Limits

Regulatory authorities set guidelines at the watershed scale but set "reduction" as a target for industry. For example, ANZECC (2000) sets trigger values for lowland rivers in south central Australia of 0.1 mg P/L, 1 mg N/L, 0.1 mg NO_x/L and 0.1 mg NH_4^+/L but does not suggest that these values should be used for individual source runoff concentrations. In WA, target nutrient losses of <0.05 kg P/ha/yr and < 0.6 kg N/ha/yr were set for the Oyster Harbour catchment but not for individual land uses in the catchment (Weaver and Prout, 1993). Dougherty *et al.* (in print) suggests that this is due to the difficulties in assessing the extent of reduction required for a particular catchment. The P sources may be highly variable in a catchment and hence it is difficult to directly link the surface water quality measured at the farm or paddock scale to the watershed.

In some states of the US, soil P concentration threshold values were set in response to an outbreak of *Pfiesteria piscidia* (a dinoflagellate) which caused neurological damage in humans (Heathwaite and Sharpley, 1999). The threshold values varied between states and depended on the analysis method recommended. The thresholds are shown in Table 3.6.

Table 3.6 Soil threshold values from some US states.

Soil test	Threshold values in soil (mg/kg)	
	Agronomic threshold*	Environmental threshold
Mehlich I	25	50
Mehlich III	30-50	130-150
Olsen	12	50-100**
Bray 1	20-40	75-150

From Heathwaite and Sharply (1999)

* based on non-vegetable crops, actual values vary with crop, these are indicative only.
** depending on soil texture, lower value for sandy soil and upper value for silty loam soil.

Management recommendations for the protection of water quality should be undertaken once the soil concentrations has equalled or exceeded environmental threshold. These recommendations include:

- Apply no more P until significantly reduced or apply limited rate if between agronomic and environmental threshold or P rate not to exceed crop removal;
- Provide buffers next to streams;
- Overseed pastures with legumes to aid P removal, provide constant cover to minimise erosion;
- manipulate P sources to reduce P solubility, including addition of alum to manure or application of soil amendments (such as fly-ash and Fe-oxides);
- Find alternative uses for manures to enable control of rate and timing of application. Consideration should be given to biosecurity concerns with transporting manures long distances (particularly for avian diseases).

The limitation of this approach is that it does not take account of the Critical Source Areas and hence valuable resources may be wasted on treating areas that are not contributing significant amounts of P to surface waters. Heathwaite and Sharpley (1999) recommend P management should focus on CSA's and N management focus on groundwater recharge areas.

3.5 Soil Treatments to Reducing Nutrient Loss

3.5.1 Reactive barriers for NO_3 remediation

Robertson *et al.* (2000) installed reactive barriers at four different sites to provide passive *in situ* treatment of groundwater. The reactive barriers were comprised of waste cellulose solids (eg. wood mulch, sawdust, leaf compost) in a blanket or a large container. Two barriers were installed horizontally and one vertically to treat leachate from septic absorption trenches and a fourth barrier was installed as a containerised reactor to treat runoff from a dairy farm.

The cellulose solids provided a carbon source for heterotrophic denitrification of nitrate as per the following equation:

$$5CH_2O + 4NO_3^- \rightarrow 2N_2 + 5CO_2 + 3H_2O + 4OH^-$$

The reaction occurs under anaerobic conditions and hence the reactive barrier must be maintained in a saturated condition. In the horizontal barriers, a fine, silty sand was mixed with 15% by volume cellulose solids to ensure the barrier remained saturated and anaerobic. The vertical wall was installed in the groundwater aquifer and hence remained anaerobic below the water table surface.

The hydraulic loading rates varied from 6 L/day to 200 L/day with the estimated retention time in the barriers as over 10 days. Input concentrations varied from 1 mg/L to 57 mg/L and the barriers reduced the nitrate concentrations by 58-91%, resulting in concentrations < 3 mg/L for three of the reactors. The reactor receiving the highest nitrate concentration (57 mg/L) reduced the concentration by 80% to 11.6 mg/L. Denitrification rates were dependent on the temperature and ranged from 1 to 32 mg L N/day and after seven years of operation remained the same as in the first year. The dissolved chloride and dissolved organic carbon concentrations were unaffected by treatment. Over the six to seven years the barriers have been operating, 2-3% of the added carbon source has been utilised.

During the trial period the effluent flow was reduced for a period of time which appeared to have resulted in the barrier drying out. During this time the output nitrate concentrations where higher but reduced again once the soil, and hence barrier, became wetter again.

Reactive barriers appear to be a viable long-term option for treatment of nitrate plumes, either in the soil or in the groundwater. Robertson *et al.* (2000) suggest that the choice of cellulose material will depend on hydraulic retention of barrier required, permeability requirements, acceptable frequency of maintenance and local availability of materials.

3.5.2 Products for Decreasing P solubility

The push to reduce waste and increasing costs of disposal have resulted in companies seeking opportunities to turn their waste into another's resource. Processing industries supporting the mining industry have a number of products that may be suitable. The following discussion is intended to provide examples of uses for products in decreasing soluble P concentrations in soil. Any wastes proposed to be applied to land should be assessed on a case-by-case basis.

Coal Technology By-products

Stout *et al.* (1999) trialled the application of coal technology by-products in laboratory conditions. The products were fly-ash from fluidised bed combustion (FBC) and gypsum from flue-gas desulfurisation (FGD). Previous research is cited as showing that application of these products has no detrimental effect when used at recommended rates.

The trials used the equivalent of 0, 5, 10 and 20 kg/t application rates. Application of FBC fly-ash or FGD gypsum at 10 kg/t reduced the soluble P concentration in the soil by 50-60%, with only slight further reductions found at the 20 kg/t application rate. The effect on soil P concentration as measured by the Mehlich III method was not as marked. Further testing of phosphorus forms showed that FBC fly-ash application transformed readily available phosphate and iron and aluminium-bound phosphates to less soluble calcium-bound phosphorus fractions, while FGD gypsum transformed the phosphorus into iron- and aluminium-bound forms.

The mechanism by which the FBC fly-ash decreased the available P was by increasing the soil pH to 8.0 and increasing the calcium content which bound the P in calcium phosphate compounds. The mechanism by which the FGD gypsum worked was by the addition of calcium, which displaced aluminium and/or iron into the soil solution, which in turn resulted in a decrease in soil pH and the production of iron- and aluminium-bound phosphates.

Modelling of two crops showed application would have an effect on the runoff concentration reaching a stream where the paddock was adjacent to the stream. Where a distance existed between the paddock and the stream, application was unlikely to have an effect.

Both the fly-ash and the gypsum waste products were able to reduce the soluble P concentration in the soil by a similar amount, though this was achieved by different mechanisms. Application of these products to field sites should consider the potential effect of increasing or decreasing the soil pH prior to application and should be targeted to critical source areas (CSAs).

Bauxite Residue (Red Mud)

Red mud is a by-product of the alumina industry, comprised of silt-sized silica and iron- and aluminium-oxides with a pH of 11 created by sodium salts. Summers *et al.* (1993) applied red mud to sandy soil of the Peel-Harvey estuary. These soil types have predominantly quartz sand with low iron and aluminium content and low P sorption capacity. Red mud was mixed with waste phosphogypsum and applied at 80 t/ha to one of a paired catchment.

The P concentration in the stream flow was reduced to < 1 mg/L in the Red Mud treated catchment compared to the untreated catchment, which had concentrations of 1-6 mg/L. The total runoff from the treated catchment was higher, which was suggested as likely to be due to decreased permeability. In addition, the treated catchment was lower lying and likely to have a higher water table which would result in runoff occurring sooner after the commencement of rainfall. The EC in the stream flow increased from < 40 mS/m in the untreated catchment to 30-80 mS/m in the treated catchment.

Application of red mud reduced the P lost from the catchment in stream flow from 13 kg/ha to 4 kg/ha, even though an additional 41 kg P/ha was applied in the phosphogypsum. However, consideration would need to given to the impact of applying sodium to the site.

Lime

The bioavailability of P in runoff is influenced by the physicochemical factors of the soil, such as loosely bound phosphate concentration, calcium carbonate, iron- and aluminium-oxides, organic carbon and clay content, and the concentration gradient between particulate and dissolved P in runoff. Sova (1996) added lime to an acidic soil to increase the pH from 4.9 to 7. One month after application, the erodibility of the soil between lime-applied and unamended soil was the same, showing that improvement in soil structural stability did not occur in this timeframe. Differences in total P measured in runoff were also not significantly different between the treatment and control and 99% of the P was as particulate P, however the form of P in the runoff was different. The P in the runoff from the acidic soil was predominantly as iron- and aluminium-bound P and the application of lime resulted in increases in the proportion of P present in loosely-bound P and decrease in iron- and aluminium-bound P.

Application of lime may result in an increase in bioavailable P in runoff sediments, where the pH of the soil is changed from an acidic soil to a neutral soil. However, Stout *et al.* (1999) showed that increasing the soil pH decreased the bioavailable P in the soil. Hence the application of lime may increase or decrease soluble P concentrations in the soil, depending on the pH of the amended soil.

Polyacrylamide Treatment for Erosion Control

High molecular weight, anionic polyacrylamide (PAM) has been applied in irrigation water to soil during furrow irrigation to stabilise soil and flocculate suspended sediment. Lentz *et al.* (1998, 2001) compared a control (untreated) furrow irrigation with PAM applied at 10 mg/L during furrow advance (i.e. until the irrigation water reached the discharge end of the furrow) and PAM applied at 1 mg/L throughout the irrigation.

The total soil losses over 4 irrigations were reduced with the application of PAM from 3 t/ha for control plots to 0.24-0.3 t/ha for PAM applied plots (Lentz *et al.*, 1998). The total and ortho-P concentrations were 5-7 times higher in control than PAM treated plots and COD was 4 times higher in the control than the PAM treated plots. PAM application increased infiltration and decreased runoff compared to the controls. Overall, PAM applied during furrow advances reduced nutrient concentrations in the runoff more than PAM applied continuously, however, due to variation in the runoff volumes from the plots where PAM was applied during furrow advance only, the cumulative loads were similar for the two application methods. The exception was for nitrate concentrations, which were not reduced by PAM application. This suggests that NO_3 is dissolved and not particulate and hence the concentrations are not affected by a reduction in sediment loads. The reduction in runoff with the PAM application would reduce the total N load lost from the soil compared to the control plot.

Further trials undertaken comparing only the control and PAM treatment in the furrow advance stage found the mean cumulative runoff sediment loss over a 12-hour irrigation was 12 kg for control furrows compared to 1 kg for PAM-treated furrow, i.e. 90% reduction in sediment (Lentz *et al.*, 2001). The PAM treatment to furrow advance compared to the control reduced the total P by 92%, molybdate reactive P by 87% and COD by 85%. These trials also found no field-wide, season-long increase in water, nitrate or chloride leaching.

Extrapolation of this overseas experience suggests polyacrylamide may be effective in reducing nutrient and sediment losses under Australian conditions. However, no trial work was found during this literature review on the potential for application of polyacrylamides to reduce nutrients in rainfall-induced runoff from Australian sites.

It is known that research on the use of polyDADMAC to reduce phosphorus in surface runoff from South Australian soil is currently underway at Adelaide University. PolyDADMAC is poly diallyldimethylammonium chloride, which is a low molecular-weight, high-charge-density and cationic compared to PAM, which is high molecular weight and non-ionic. Both are referred to as homopolymers and are water soluble and used as flocculants to remove clays, organo-mineral complexes etc. Sometimes they are grafted together using glycerol as the plasticiser or gamma radiation. The copolymer improves flocculation and sludge dewatering performance.

The work on polyDADMAC is in its infancy compared with PAM and started being referred to in the literature approximately 5 years ago.

This technology is evolving and may have further application in the future but the spreading of polyacrylamides to reduce nutrient concentrations can not be promoted as a management option until the findings of this research are known.

3.6 Summary

Nutrient movement may be managed by:

- choosing an appropriate wastewater treatment system;
- limiting the density of systems in a catchment area;
- selecting a suitable site for effluent application;
- targeting Critical Source Areas (CSAs) and groundwater recharge areas for specific management strategies;
- managing the disposal area to maximise nutrient uptake by plants, gaseous loss to the atmosphere and sorption to soil organic matter and exchange complexes.
- minimising runoff through crop and land management;
- utilising soil treatments to increase gaseous losses or decrease nutrient solubility.

4. Separation Distance Review

Separation distance (or setback) is the linear distance between an activity and an identified sensitive area   Separation distances are also referred to as buffer distances or buffer zones, though the latter term denotes more the area prescribed by the separation distance. Vegetated filter strips are the area prescribed by the separation distance and where vegetation is fully established in this area.

4.1 South Australian Separation Distance Guidelines

The SA EPA has "Draft Guidelines for Separation Distances" (SA EPA, 2000), which were released for public comment in August 2000. These guidelines have not been finalised but may still provide guidance on acceptable separation distances. The SA Health Commission recommended separation distances from waterbodies for liquid waste disposal are:

- 50 m from any well, bore, dam, or watercourse used for domestic water supply;
- 100 m from pool level for the River Murray and Lakes;
- 100 m from any well or bore and not on land subject to waterlogging or flooding for intensive animal waste disposal areas, where the waste is disposed of on the property and the manure is be spread evenly over pasture.

SA Govt (2002) includes separation distances for watercourses in its modification of Hird *et al.* (1996) limitation table. These distances are shown in Table 4.1. The inherent assumption in this table appears to be that there are no erosion and sediment control measures in place and that effluent irrigation within 50 m of a watercourse is not allowed under any circumstances. There is no mention of where the buffer zone should be vegetated and to what extent.

Table 4.1   Limitation for effluent application near watercourses.

Feature	Limitation			
	Slight	Moderate	Severe	Comments
Separation distance (m) to water course	> 200	100-200	50-100	Contamination by runoff

The Australian Standard for on-site wastewater management (AS/NZS, 2000) concentrates on design, construction and operation and maintenance of on-site systems and does not recommend separation distances for surface water or groundwater.

4.2 Recommended Separation Distances from Other States and Overseas.

A limited review of the recommended separation distances from other Australian states and overseas has been undertaken. A summary of the findings for separation distances around surface water is presented in

Table 4.2 and for groundwater is presented in Table 4.3. Surface water includes pond, lake, wetland, or sinkhole, perennial and intermittent streams.

In all cases little or no explanation or justification for the recommended distances were given. Geary (2003) notes that setbacks may be based on modelling and risk assessment (such as undertaken by Yates and Yates, 1989) although are often arbitrarily chosen due to a need to set a buffer.

In Western Australia, a stakeholder involvement programme was established to refine a set of draft site selection criteria (including separation distances) for a proposed hazardous waste facility (3C, 2004). The consultative process undertaken involved representatives from the community and industry to provide comment, all of which appear to have been based on perception and not on science as were highly disparate. For example, the suggested separation distances for protection of natural waterways was 1000 m for high conservation/ ecological value systems, 500 m for slightly to moderately disturbed ecosystems and 250 m for highly disturbed ecosystems. Suggestions from participants were to increase the separation distance on highly disturbed systems to up to 1000 m and some believed that a distance of 10 km was more appropriate for aquatic systems with high value.

Others have suggested that separation distances are set based on the smallest distance that can possibly be defended and not on protection of the waterway (VPIRG, unknown).

Recommended buffer distances may also be related to slope. Smith (1992) presents minimum recommended widths of vegetated buffer zones from the Soil Conservation Service Field Office Technical Guide (1988) that are based on the slope of the land, such that:

- 0-10% 5 m vegetated buffer zone;
- 10-20% 6 m vegetated buffer zone;
- 20-30% 8 m vegetated buffer zone.

Table 4.2 Summary of separation distances of effluent and manure application areas from surface water from other states and overseas.

Location	Reference	Waste type	Separation distance (m)	Situation	Conditions
NSW	DLP *et al.* (1998)	sewage effluent	100	permanent source	Based on ideal site and soil conditions. Major limitation if located below 1:20 yr flood contour and suggests transport of wastewater off-site
			40	farm dams, intermittent waterways and drainage channels	Based on ideal site and soil conditions
Queensland	DNRM (2002)	sewage effluent	50	primary effluent	No requirements for flood-prone areas
			30	secondary effluent	
			10	advanced secondary effluent	
Tasmania	Sorrell Council (undated)	sewage effluent	10-100		depending on slope, soil permeability, type of land application system, nature of buffer zone, sensitive features, effluent quality and climate. No requirements for flood-prone areas
Victoria	VicEPA (2002)	sewage effluent	300	potable water in dam or reservoir	may be reduced by up to 50% if effluent quality is high, local council has a compliance program and the adjacent slope is < 5%
	VicEPA (2002, 2003)		100	Potable water supply catchment streams	may be reduced by up to 50% if effluent quality is high, local council has a compliance program and the adjacent slope is < 5%. High risk for land with < 1:20 yr flood return Very high risk for land with <1:10 yr flood return
			60	other surface water	may be reduced by up to 50% if effluent quality is high, local council has a compliance program and the adjacent slope is < 5% High risk for land with < 1:20 yr flood return Very high risk for land with <1:10 yr flood return
Western Australia	DA *et al.* (2002); Latto *et al.* (2000)	cattle and pig effluent	100	permanent river or stream	above 1:100 yr flood level Piggeries should not be established in catchment of any wetland or water body with aesthetic, scientific, cultural or recreational significance.
			50	intermittent water courses	
			200	wetlands	measured to boundary of wetland vegetation around estuaries and lakes.
Texas, Virginia and Pennsylvania	USEPA (2000)	sewage effluent	7.5-30		
Alabama	USDA (2001)	Animal manure	15	General waters where manure	Cultivated land for waste application must have adequate erosion control

Location	Reference	Waste type	Separation distance (m)	Situation	Conditions
	LPES (2001)			applied to hayland, pasture or cropping land	Vegetated filter > 15 m wide of permanent grasses with a stem density > 1550 stems/m^2. If incorporates a riparian forest buffer (in accordance with NRCS Conservation Practice Standard, Riparian Forest Buffer – Code 391A), the permanent grass filter strip may be 6 m wide. Vegetated width must be located adjacent to the application field or concentrated flow area and be shaped so that flow from runoff is uniform (sheet flow) and does not concentrate.
			60	sensitive	Public water supply or outstanding water resource
Arkansas	LPES (2001)	manure	30		
California	LEPS (2001)	manure	no requirements		
Alabama			15		
Georgia	LEPS (2001)	manure	30	1000-3000 animal units	
			45	> 3000 animal units	
Illinois	LEPS (2001)	manure	60		
Indiana	Frankenberger *et al.* (unknown)	Feedlot manure	8	Liquid manure	Injection or single pass incorporation Reduced to 1.5 m if a drainage inlet
			15	Liquid or solid manure	incorporation or pasture application
			30	Liquid manure	Surface application on < 6 % slope or with crop residue
			60	Liquid manure	Surface application > 6% slope
Iowa	Iowa DNR (2003) LEPS (2001)	Feedlot manure	0	dry or liquid manure	surface application and incorporation within 24 hours or injection
			15	dry or liquid manure or irrigated	buffer area vegetated incorporation > 24 hours or not at all and low or high pressure irrigation
			60	dry or liquid manure or irrigated	buffer area unvegetated incorporation > 24 hours or not at all and low or high pressure irrigation
			250	dry or liquid manure	High quality water resource Buffer area unvegetated Incorporation > 24 hours or not at all
			250	Irrigation	High quality water resource Low or high pressure irrigation
Michigan	Frankenberger *et*	Feedlot manure	50	surface water or areas subject to	Inject or surface apply and incorporate within 48 hrs

Location	Reference	Waste type	Separation distance (m)	Situation	Conditions
	al. (unknown)			flooding	Use conservation practices to protect against runoff and erosion
Minnesota	LEPS (2001)	manure	15		
			8-90	"special" practices	these were note defined
Missouri	Missouri DNR (2004)	animal manure	30	permanently flowing streams	
			15	intermittently flowing streams	
			90	other surface water	
Kansas	LEPS (2001)	manure	30		general recommendation for all site features
New Mexico	LEPS (2001)	manure	30		
North Carolina	LEPS (2001)	manure	15		
Oklahoma	LEPS (2001)	manure	15	intermittent	
			30	perennial	
			60	perennial	slope > 8%
Pennsylvania	LEPS (2001)	manure	30	above application area	
			60	below application area	
Texas	LEPS (2001)	manure	30		
Virginia	LEPS (2001)	manure	15		unincorporated
			7.5		incorporated

Table 4.3 Summary of separation distances of effluent and manure application areas from groundwater from other states and overseas.

Location	Reference	Waste type	Separation distance (m)	Situation	Conditions/Comments
NSW	DLP *et al.* (1998)	sewage effluent	250	Domestic groundwater well	Based on ideal site and soil conditions
Queensland	DNRM (2002)	sewage effluent	50	Used for human consumption	Primary treatment of effluent
			30	Used for human consumption	Secondary treatment of effluent
			10	Used for human consumption	Advanced secondary treatment of effluent
Tasmania	Sorrell (undated)	sewage effluent	variable	Used for human consumption	Determined by groundwater geologist using model
			1	Water table	Below base of trench. Lesser distance for secondary treated effluent. Greater distance required in some soils and trenches may not be permissible if groundwater pollution may occur.
Victoria	VicEPA (2002)	sewage effluent	20	Potable or non-potable bore	may be reduced by up to 50% if effluent quality is high, local council has a compliance program and the adjacent slope is < 5%
Western Australia	Latto *et al.* (2000)	animal effluent	100	Bores for potable or farm supply	May be increased to 300 m if effluent is raw or partially treated.
	Latto *et al.* (2000); WABGA and PFAWA (2004)		1.5-2	Water table	Depending on industry Buffer to be vegetated
Texas, Virginia and Pennsylvania	USEPA (2000)	Sewage effluent	30	Wells	
Alabama	USDA (2001)	Animal manure	60	Well upgradient	
			90	Well downgradient	
	LPES (2001)		30	Non potable	
Arkansas	LPES (2001)	Animal manure	30		
California	LEPS (2001)	Animal manure			no requirement
Georgia	LEPS (2001)	Animal manure	30	1000-3000 animal units	
			60	> 3000 animal units	
Illinois	LEPS (2001)	Animal manure	45		

Location	Reference	Waste type	Separation distance (m)	Situation	Conditions/Comments
Indiana	Frankenberger *et al.* (unknown)	Feedlot manure	150	public wells and intake structures	
			15	private wells	liquid incorporated
			30	private well	surface application on < 6 %
			60	private well	surface application on > 6 % slope
Iowa	Iowa DNR (2003)	Feedlot manure	0	ag drainage well and dry or liquid manure	surface application and incorporation within 24 hours or injection
			60	ag drainage well and dry or liquid manure	incorporation > 24 hours or not at all and low or high pressure irrigation
			Not allowed	Irrigation	
Kansas	LEPS (2001)	Animal manure	30		general separation distance requirement
Minnesota	LEPS (2001)	Animal manure	15		
Missouri	Missouri DNR (2004)	Animal manure	90	Wells	
New Mexico	LEPS (2001)	Animal manure	60	public well	
			30	private well	
North Carolina	LEPS (2001)	Animal manure	30		
Oklahoma	LEPS (2001)	Animal manure	No requirements		
Pennsylvania	LEPS (2001)	Animal manure	30		
Texas	LEPS (2001)	Animal manure	150	public well	
			45	private well	
Virginia	LEPS (2001)	Animal manure	30		

4.3 Effectiveness of Vegetated Buffer Zones

Vegetated buffer zones decrease the runoff flow hydraulics and hence allow time for infiltration of water, deposition of sediment and adsorption of nutrients. The total sediment decrease through a vegetated buffer was 60-90%, regardless of whether the buffer was grass or mixed riparian vegetation (Daniels and Gilliam, 1996). Variation in sediment reduction was related to whether the watershed had been recently cultivated and the storm intensity. High flow volumes may overwhelm grass and riparian buffer zones and reduce their effectiveness.

Reduction in nutrients is dependent on the form of the nutrient in the runoff. Total P may be reduced by 50% but 80% of soluble P frequently passes through the buffer zone (Daniels and Gilliam, 1996). Buffer zones may also reduce ammonia by 20% to 50% and TKN and NO_3 by 50%, with NO_3 reduction by denitrification dependent on the moisture conditions of the buffer zone. A grass buffer receiving pig effluent at the upslope end removed 44% of applied N, 19% of applied P and 23% of applied potassium (K) as grass biomass alone (Hubbard *et al.*, 2003).

Nash and Murdoch (1997) note that buffer strips are unlikely to be useful in pastured watersheds as runoff only occurred during saturated conditions and hence strips would only affect settling. Due to the low settling velocity of sediment sizes < 0.45 µm, this would result in the majority of sediment and nutrients moving through the buffer zone.

4.4 Modelling

Cromer *et al.* (2003) suggests that fixed separation distances are untenable on a large scale where this is over disparate climatic and soil conditions. Modelling may provide an alternative method of determining separation distances. A number of models are mentioned by Geary (2003) that can predict pathogen die-off, virus fate and estimate impacts from nitrogen. Geary (2003) found through modelling that viruses could potentially move to waters through soil even though adhering to recommended separation distances.

Yates and Yates (1989) used disjunctive kriging to calculate separation distances and the probability that a defined separation distance was adequate to prevent viral contamination of groundwater at the setback boundary. This modelling suggested that the there was a 70% probability that 15m was an adequate separation distance. The probability increased to 85% when the separation distance was increased to 30 m. A 40 m separation distance would be required to achieve a probability of 90% and an 80 m separation distance would be required to achieve a probability of 99%. It should be noted that this method of modelling would require expert input.

Charles *et al.* (2003) are reviewing processes undertaken to develop performance-based buffers for Sydney's drinking water catchments, including the use of risk assessment techniques to the application of pathogen and nutrient fate and transport modelling. A model would need to include contaminant transport and fate factors, including surface runoff (rainfall and slope), soil saturation and depth and groundwater heterogeneity and flow, buffer distance slope, vegetation and heterogeneity, and the contaminant fate in the catchment, which is controlled by stream water quality.

4.5 Summary

Limited justification was given for separation distances set by Authorities in Australia and the United States. Separation distances for surface water vary from 0 m to 300 m depending on the waste applied, the type of surface water and the site management practices employed. Similarly, separation distances from groundwater vary from 1 m to 250 m depending on the use of the well, the quality of the effluent and the method of application.

The requirements for the type of buffer also vary with not all Authorities requiring the buffer zone to be adequately vegetated. Although the effectiveness of vegetation has been found to be limited some cases, most research has suggested some, if minor, reduction in nutrients leaving the buffer zone is achieved.

Modelling of the nutrient movement is likely to be prohibitively expensive in many circumstances and may not be adequately definitive. However, this technology may be required where the site has moderate or severe limitations or effluent irrigation within a defined buffer is proposed.

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