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Phosphorus sustainability during irrigation

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Contents

List of figures

List of tables

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Executive summary

Eutrophication, the accumulation of nutrients such as nitrogen, phosphorus and carbon in streams and water impoundments is a worldwide problem (Cooke et al., 1993; USEPA, 1996; European Environment Agency, 1998). Sites where abattoir wastewaters are applied to land are one possible source of the phosphorus adversely affecting our water supplies.

For phosphorus exports to be a problem at a wastewater irrigation site there needs to be (1) a source of phosphorus that is (2) mobilised and (3) transported to a location where (4) its adverse effects are expressed, often through a vector such as blue-green algae. Abattoir wastewaters vary depending on a range of factors including the type of animals being processed and wastewater treatment. In general abattoir wastewaters contain significant quantities of salt, phosphorus, nitrogen and labile organics measured as the biological oxygen demand (BOD), and have a high sodium adsorption (SAR) ratio. All these components have the potential to affect phosphorus exports. It is of note that as a result of treatment costs, the United States Environment Protection Agency did not mandate treatment of abattoir wastewaters for phosphorus in its recent Final Effluent Limitations (Environment Protection Agency, 2004).

The source of phosphorus exported from wastewater application sites may be the wastewater itself, plants, soil, and/or plant and animal waste products. Plant roots take up inorganic phosphorus in the form of orthophosphate from soil water and, accordingly, the reactions of inorganic phosphorus in soil have been widely studied. Precipitation/dissolution and adsorption/desorption processes maintain the concentration of inorganic phosphorus in soil water, the latter being more important at the concentrations expected at most wastewater irrigation sites. In Australian soils, adsorption/desorption reactions generally occur at the surface of iron and aluminium oxides. The ability of the soil to adsorb phosphorus is commonly called the soil buffering capacity and is affected by organic matter that, amongst other things, blocks adsorption sites and lowers the activity of iron and aluminium. In addition, labile organic matter added to the soil with wastewater can induce anaerobic soil conditions that result in the reduction of Iron (III) to Iron (II), affecting phosphorus adsorption. Anaerobic soil conditions are more likely to develop where the infiltration capacity of the soil has been impaired by the accumulation of suspended solids or by the adsorption of excessive sodium to soil colloids.

Organic forms of phosphorus have received far less study than inorganic forms. Strong evidence that organic phosphorus contributes directly to phosphorus exports can be found in the many studies where organic and inorganic phosphorus in drainage have been compared.

Phosphorus is mobilised by two processes: (1) physical detachment of sediment (>0.45μm) and the associated phosphorus (often called erosion) and (2) dissolution. Detachment is essentially a physical process that is largely driven by the kinetic energy of the water. Dissolution is affected by the chemistry of phosphorus and surrounding materials and the available reaction time. Dissolution is the most important mobilisation process at well managed wastewater irrigations sites.

Once mobilised, phosphorus can be transported offsite through surface or sub-surface pathways. Surface pathways include infiltration excess and saturation excess overland flow and near surface interflow. Sub-surface pathways include vertical matrix flow and macropore flow. The concentrations of phosphorus transported through the different pathways depend on the time available for adsorption/desorption reactions to occur with highest concentrations usually found in overland flow followed by macropore flow and vertical matrix flow.

The complexity of biological systems makes it difficult to develop definitive regulations that enable the application of abattoir wastes to land without excessive phosphorus exports. In most countries it would appear that the application of abattoir wastes to land has been regulated on a case by case basis often using nitrogen, rather than phosphorus, as the limiting nutrient. In the United Kingdom blood and gut contents from abattoirs have an exemption from the need for a license under the Waste Management Licensing Regulations (1994). In the Netherlands it was difficult to find information specifically relating to abattoirs. While farm nutrient budgeting is

mandated, it was not clear if these rules applied to abattoir wastes or whether such wastes were supplied to the manure trading system. In the United States it would appear that abattoirs are regulated on a case by case basis despite phosphorus indices having been developed for the application of phosphorus to agricultural systems over most of the country.

In Australia, the regulation of abattoir wastewater applications to land is undertaken on a case by case basis and is framed in terms of protecting environmental assets. It is clear from the information collected that regulation varies both between and within states. In Victoria, licenses can specify maximum phosphorus concentrations. However, in at least some cases these are likely to be exceeded on a regular basis. Most States had general guidelines for the management (i.e. loading rates) of abattoir wastewater irrigation sites but these guidelines were not enforceable.

Almost without exception, the Australian Government agencies noted that there was a trend towards the utilisation of the nutrients contained in abattoir wastewaters rather than disposal. As a result, increased emphasis was being placed on the development of nutrient management plans, especially for new sites. The South Australian Government has developed a spreadsheet that is being used for monitoring sites. The phosphorus component of the spreadsheet contains a target Phosphorus Retention Index measure (0.5 mg P/L) that, when exceeded, requires that phosphorus inputs and outputs are balanced.

There are a number of soil tests that could be used to monitor abattoir wastewater application sites. These test generally fall into three categories; (1) agronomic soil tests, (2) adsorption/desorption tests and (3) composite soil tests. Agronomic soil tests, such as Olsen P and Colwell P, have been used extensively to monitor wastewater application sites. Such tests use extractants such as bicarbonate to predict phosphorus availability over the growing season of a crop. For environmental testing there is an increasing trend towards the use of distilled water or dilute electrolyte solutions and short extractions periods to better reflect phosphorus sources that may be mobilised in storm periods. Adsorption/desorption tests have been used for experimental purposes over an extended period. They estimate the ability of the soil to hold phosphorus and when modelled mathematically can be incorporated into numerical models. Composite soil tests are increasingly being used for environmental purposes. They relate the concentration of phosphorus extracted from soil to other chemical species, most commonly iron and aluminium in the same solution, or to other tests.

Soil phosphorus tests are an invaluable tool for estimating the source component of phosphorus exports. However, the value of any soil test in predicting phosphorus export potential is limited to the system under investigation, the depth at which the soil is collected and the primary pathway for phosphorus exports. In isolation, soil tests are a very poor estimate of phosphorus export potential.

In the United States, index systems that combine quantitative (i.e. soil test data) and qualitative information relating to phosphorus sources, mobilisation and transport have been used to assist with the management of phosphorus applications to farmland. While these indices are a simplistic representation of complex biological systems, they are useful because they give general guidance for management and the changes that are necessary to protect water quality, rather than attempt to quantify phosphorus exports.

Numerical models are used for managing wastewater generation and irrigation systems. MEDLI is an example of such a model developed by the CRC for Waste Management and Pollution Control and the Queensland Departments of Natural Resources and Primary Industries. MEDLI models effluent production, its treatment and ultimate disposal on land and predicts the fate of water, nitrogen, phosphorus and soluble salts. Consequently, MEDLI is used for analysing and designing effluent disposal systems for intensive rural industries using land application and would appear to be extremely useful in that regard.

In order to retain its simplicity and flexibility MEDLI uses a phosphorus adsorption isotherm (single Freundlich form) to predict the phosphorus in drainage. As a result MEDLI is likely to under-predict the capacity of the soil to store phosphorus and overestimate phosphorus in drainage. These simplifications would appear to be major limitations if such a model were to be used for regulatory purposes (i.e. setting maximum phosphorus loadings).

An alternative modelling framework that could be used to develop a set of rules on which regulation could be based is Bayesian Networks. Bayesian Networks incorporate the simplicity and transparency of phosphorus indices and the complexity of numerical models. While Bayesian Networks only produce quasi-quantitative numerical values (i.e. based on the probability distributions of nodal classes), their value lies in the quantification of causal relationships and their ability to identify the most sensitive factors affecting the output.

Given the deficiencies of the current range of tools available to Australian regulators, it would appear that a national project aimed at developing a conceptual model of phosphorus reactions and exports from abattoir wastewater irrigation sites is warranted. An initial model based on a Bayesian Network could be updated using information collected as part of the regular monitoring that is required of most wastewater irrigations sites in Australia. When sufficiently robust, the Bayesian Network could form the basis of the phosphorus algorithms in numerical models such as MEDLI.

Introduction

Eutrophication, the accumulation of nutrients such as nitrogen, phosphorus and carbon in streams and water impoundments is a worldwide problem (Cooke et al., 1993; USEPA, 1996; European Environment Agency, 1998). The adverse effects of eutrophication include excessive growth of weeds and algae, and oxygen depletion that results from their death and decomposition. These can restrict the use of key water resources for fisheries and recreation, and as industrial and domestic water supplies. In addition, eutrophication contributes to the recurring and often explosive growth of cyanobacteria (also called blue-green algae) in reservoirs used for domestic water storage. Such blooms cause fish kills, reduce the palatability of the water and lead to the formation of trihalomethanes during water treatment (Kotak et al., 1993). In addition, endotoxins, hepatotoxins and neurotoxins that are produced by some cyanobacteria can kill domestic stock and pose a health risk to humans (Department of Natural Resources and Environment, 1996).

Much of the eutrophication we now observe is the result of human activities (Carpenter et al., 1998). Of the nutrients contributing to this accelerated or cultural eutrophication (Department of Natural Resources and Environment, 1996), considerable attention has focused on phosphorus. While carbon and nitrogen are also essential for the growth of aquatic organisms, access to them is difficult to control in an aquatic environment because many organisms can access atmospheric sources (Sharpley and Rekolainen, 1997).
For many years Australia, like the rest of the world, has suffered the adverse impacts of eutrophication. For example, in 1878 a toxic outbreak of *Nodularia spumigena* occurred in Lake Alexandrina in South Australia (Francis, 1878). Major algal blooms have since occurred in all the mainland states and appear to have resulted in both human illnesses and stock deaths (Department of Natural Resources and Environment, 1996). In late 1991 the Darling-Barwon system recorded what was at the time the largest riverine algal bloom in the world, extending for 1000 km, with an estimated cost of $1.3B AUS (Department of Water Resources, 1992).
Wastewater irrigation sites are one possible source of the phosphorus adversely affecting freshwater systems. For phosphorus exports to be a problem at a wastewater irrigation site there needs to be (1) a source of phosphorus that is (2) mobilised and (3) transported to a location where (4) its adverse effects are expressed, often through a vector such as blue-green algae (Nash, 2002). While their catchment-scale impact will vary with location and wastewater characteristics, the preference for wastewater reuse on land rather than disposal to water is likely to increase the pressure on land application systems to demonstrate their environmental sustainability.

Meat and Livestock Australia provides research and development, market information and marketing for the benefit of Australia's red meat industry. As part of their commitment to protecting Australia's natural capital, Meat and Livestock Australia has asked the Victorian Department of Primary Industries through its research arm, Primary Industries Research Victoria (PIRVic), to review tools that could be used to predict the offsite impact of the phosphorus in meat processing wastewaters applied to land.

In this report to Meat and Livestock Australia we:

- Review abattoir wastewater characteristics;
- Use the review of abattoir wastewater characteristics to discuss the sources, mobilisation and transport of phosphorus in wastewater irrigation systems with particular emphasis on wastewaters such as those from abattoirs that generally have a high biological oxygen demand (BOD) and salt content (TDS);
- Review the current approaches to calculating/licensing phosphorus loadings when meat processing wastewaters are applied to land; and
- Critically review different approaches/tools used for predicting the offsite impact of phosphorus where meat processing wastewaters are applied to land.

Abattoir wastewater characteristics

A summary of wastewater characteristics from published literature is presented in Table 1. It is notable that these wastewaters contain variable but generally significant quantities of salt, phosphorus, nitrogen and labile organics as measured by the biological oxygen demand (BOD) and chemical oxygen demand (COD). In addition, the pH varies over a wide range, presumably as a result of water treatment and the use of caustic cleaners, and the wastewaters generally have a high sodium absorption ratio (SAR).

Table 1. A summary of wastewater characteristics[a] from published literature.

a) General characteristics

TSS (mg/L)	Fat (mg/L)	TKN (mg/L)	NH_4^+ N (mg/L)	NO_x (mg/L)	COD (mg/L)	BOD_5 (mg/L)	TP (mg/L)	SAR	pH	EC (mS/cm)	
829	317	391	169	98	5559	1395	36	11	7	7	*Mean*
		5	2		1.9	1.2	0.5	7	0.5	1.7	*Min*
6300	1200	16500	3500	960	150000	10000	183	13.9	11.4	14.8	*Max*

b) Selected inorganic ions

Cl (mg/L)	Na (mg/L)	K (mg/L)	Ca (mg/L)	Mg (mg/L)	Mn (mg/L)	
426	250	94	60	9	0.38	*Mean*
4.8	0.1	20	3	3	0.38	*Min*
2200	2000	798	300	27	2	*Max*

Sources: (Cooper et al., 1979; Keeley and Quin, 1979; Weber and Hull, 1979; Burden, 1984; Lovett et al., 1984; Russell et al., 1984; Russell, 1986; Davies and Payne, 1988; Tiller, 1988; Coulliard et al., 1989; Quinn and McFarlane, 1989; Metzner and Temper, 1990; Russell et al., 1991; Bowmer and Laut, 1992; Hansen and West, 1992; Russell and Cooper, 1992; Sangodoyin and Agbawhe, 1992; Tritt, 1992; Dimitriov and Russell, 1993; Subramaniamd et al., 1994; Benka-Cocker and Ojior, 1995; Borja et al., 1995; Page et al., 1997; Masse and Masse, 2000; Guo and Sims, 2001; Webb and Ho, 2001; Caixeta et al., 2002; Adeleye and Adebiyi, 2003; Al -Mutairi et al., 2004; Filali-Meknassi et al., 2004; Godlinski, et al., ; Luo, Lindsey et al., 2004)

[a] TSS, Total Suspended Solids; TKN, Total Kjeldahl Nitrogen; NH_4^+ N, Ammonium Nitrogen; NO_x, Nitrate /Nitrite; COD, Chemical Oxygen Demand; BOD_5, Biological Oxygen Demand; TP, Total Phosphorus; SAR, Sodium Adsorption Ratio; EC, Electrical Conductivity; Cl, Chloride; Na, Sodium; K, Potassium; Ca, Calcium; Mg, Magnesium; Mn, Manganese.

Sources of phosphorus at wastewater irrigation sites

Introduction

In agricultural systems, the plants, animals and soil are potential stores and sources of phosphorus. Plants assimilate inorganic phosphorus from the soil solution. Grazing animals assimilate phosphorus from live plants. Soil fauna and microorganisms assimilate phosphorus from live plants, dead plant and animal materials and from the soil solution. The assimilation of phosphorus into biomass exemplifies the conversion of inorganic phosphorus, largely orthophosphate, to organic forms that are unavailable to plants (i.e. immobilisation). Ultimately the organic phosphorus returns to the soil as detrital material and the organic phosphorus is converted back to orthophosphate (i.e. mineralisation).

The form in which phosphorus is stored is important when considering mobilisation and transport processes. Both organic and inorganic phosphorus can be mobilised by water directly from source materials. However, the chemical species within each group vary and that affects both in their solubility and reactivity with soil. For example, inorganic phosphorus in the form of orthophosphate can be removed from the soil solution, and therefore rendered unavailable to water (Engelstad and Terman, 1980), through precipitation reactions or adsorption onto soil particles (collectively termed fixation). As a consequence the availability of inorganic phosphorus depends on equilibrium concentrations of reactants such as iron and aluminium and the chemical conditions that influence them such as pH and oxygen potential. Organic phosphorus on the other hand, includes a wide range of compounds that are not immediately available to plants, may not be fixed in soil and are generally transformed through microbial processes. Conditions that favour microbial growth may increase or decrease organic phosphorus availability. It is of note that in addition to affecting phosphorus mobilisation and transport, the form of phosphorus also affects the impact of phosphorus on receiving waters (Sharpley and Smith, 1992).

Rainfall

When rain falls, it washes over the plants onto the soil surface. The rainwater, plants, animal waste products, the soil fabric and wastewater residuals all contribute to the nutrients contained in the water. Apart from some notable exceptions (Greenhill et al., 1983b; Sharpley et al., 1985b), it is generally accepted that rainfall makes little direct contribution to the phosphorus exported in water (Greenhill et al., 1983a).

Wastewater

Abattoir wastewaters can contain significant concentrations of total phosphorus (Table 1). However, there is little information in the literature on the relative proportions of the different phosphorus forms (i.e. organic or inorganic) and chemical species. Presumably the proportions vary with the type of animals being processed and the wastewater treatment. It is therefore not possible to draw general conclusions regarding the direct contribution of wastewater phosphorus to the phosphorus that may be exported from wastewater irrigation sites, especially since the phosphorus concentration of irrigation water is not always related to phosphorus concentrations in drainage (Nash and Clemow, 2003). It is similarly difficult to draw general conclusions regarding the indirect impact of wastewater phosphorus on phosphorus exports.

The volume of wastewater applied at an irrigation site, and therefore the loading of phosphorus, will depend on, amongst other things, the evapotranspiration potential and acceptable environmental loadings (Overcash and Pal, 1979; Nash, 1984). As an illustration, consider an area such as the Macalister Irrigation District of south-eastern Australia. In this region there are commonly 12-14 irrigations per year based on a 50 mm water deficit (Nash et al., 2003). With a phosphorus concentration in irrigation water of 36 mg P/L (Table 1), the annual phosphorus load would be ca. 270 kg P/ha. The phosphorus fertiliser application rate for this area is approximately 35 kg P/ha. Such calculations suggest that phosphorus in wastewater may ultimately increase soil fertility and therein increase phosphorus exports compared to freshwater irrigation. However, the extent to which this occurs, if at all, will depend on the system in question.

Plants

Model studies and decomposing materials have generally been used to investigate the direct contribution of plants to phosphorus export (Timmons et al., 1970; Schreiber, 1985; Schreiber and McDowell, 1985; Havis and Alberts, 1993). For example, phosphorus mobilised from ‘hayed-off’ phalaris and clover plants has been studied under a wide range of laboratory conditions (Jones and Bromfield, 1969; Bromfield and Jones, 1972). Of the total phosphorus in plant material, 60-83% was water soluble and up to 62% of this phosphorus, 37-51% of the total, was leached by 125 mm of simulated rainfall over 96 hours. The concentrations of phosphorus in leachate varied from 2-150 mg/L and the results were clearly affected by environmental conditions.

The range of soluble phosphorus found by Bromfield and Jones (1972) and phosphorus concentrations of plants grown in high and low fertility dairy pastures in the Gippsland region (Department of Natural Resources and Environment, 2001) indicate that plants produce 21-56 kg/ha of water-soluble phosphorus annually. This strongly suggests that plants may be an important source of phosphorus exported from wastewater irrigation sites, especially those with highly fertile soils. Laboratory studies (Jones et al., 1969; Bromfield et al., 1972) have shown that plants grown at higher soil solution phosphorus concentrations contain more total phosphorus, a higher percentage of phosphorus in a water soluble form, and lose about the same percentage of water soluble phosphorus as those grown in lower levels of phosphorus.

Studies of growing plants suggest that the phosphorus extracted by water depends on the plant species (Sharpley, 1981) and their pre-treatment. For example, phosphorus exported in overland flow from flood irrigated bays has been compared after mowing and grazing (Nexhip et al., 1997). Immediately after defoliation, the phosphorus exported from the mown control was greater than from the two lowest grazing pressures of 100 and 200 cows/ha over a 24 hour period. Presumably the physical damage caused by mowing was greater than the combined effects of grazing and defecation.

Soil

The surface (0-20 mm) is the most phosphorus rich zone of all but a few soils. It is here that wastewaters are applied, animals defecate, and plants deposit litter containing phosphorus that has been extracted from lower in the profile. It is here too that rainfall and soil interact, usually to a depth of only a few millimetres (Sharpley et al., 1981; Ahuja, 1986).

Soil is an extremely complex material and surface soils in particular contain a vast array of inorganic and organic compounds, detrital material, flora and fauna. Some of the forms of phosphorus in soil in decreasing order of lability include (Holford, 1989):

- phosphorus in solution (<1% of total);
- inorganic phosphorus in plant and microbial residues;
- inorganic phosphorus adsorbed on clay and organic matter surfaces;

- inorganic phosphorus ranging from sparingly soluble to extremely soluble and derived from both inorganic and organic sources;

- inorganic phosphorus occluded or absorbed in phosphorus reactive minerals; and

- very stable organic phosphorus in plant, animal and microbial material.

Such lists tend to underestimate the role of organic phosphorus in soil processes. While organic phosphorus concentrations are highly variable, they can account for 50-80% of total soil phosphorus (Richardson, 1994), especially near the surface (Perrott and Sarathchandra, 1989; Perrott et al., 1990). At sites where abattoir and similar wastewaters containing significant quantities of organic material are applied to land, soil organic phosphorus would initially be expected to be at the high end of that range. However, the proportion of inorganic phosphorus is likely to increase with time as organic phosphorus is mineralised.

Inorganic phosphorus

Plants absorb phosphorus as inorganic ions from the soil solution (Holford, 1997). Consequently, inorganic phosphorus compounds and their reactions in soil have been extensively studied (Smith, 1965; Sample et al., 1980; Barrow, 1989a; Holford, 1989) and reviewed (Wild, 1949; Norrish and Rosser, 1983; Haynes, 1984; Sanyal and DeDatta, 1991; Schulthess and Sparks, 1991).

The concentration of inorganic phosphorus in the soil solution is maintained by precipitation/dissolution and adsorption/desorption processes. Precipitation/dissolution processes are prominent at high phosphorus concentrations, for example in the soil surrounding fertiliser granules (Barrow, 1989b). Adsorption/desorption processes also occur in the immediate vicinity of fertiliser granules. However, as the wetting front progresses through the soil and phosphorus concentrations decrease, adsorption/desorption processes become increasingly important in buffering solution phosphorus concentrations (Barrow, 1989b).

Adsorption/desorption reactions in soils can be the result of fixed or variable charge. Fixed charges result from isomorphic substitution within a crystal lattice such as occurs with aluminium substitution for silicon in some clay minerals (van Olphen, 1977). Compensating cations are adsorbed to the negatively charged faces of these crystals. Variable charge can result from incomplete bonding on the edges of clay minerals and the surface of oxides, especially aluminium and iron. Water molecules, which gain or lose protons depending on the soil pH, are attracted to these surfaces. Consequently, the charge on these, and some forms of soil organic matter that can also gain and lose protons, varies with pH. It is the variable charge, especially that associated with iron and aluminium oxides, that is important for phosphate adsorption (Barrow, 1980).

Phosphate anions can replace the water and hydroxyl ions adsorbed to variable charge sites on aluminium and iron oxides. The majority of these sites are negative at the pH of most soils (Barrow, 1990). Despite also having a negative charge, phosphate ions with sufficient activation energy are able to approach the crystal surface close enough to establish short-range chemical bonds. As phosphate ions are adsorbed the surface charge becomes increasingly negative, lessening the probability that subsequent ions will be adsorbed and, in the short term (i.e. days), quasi-equilibrium is established with the soil solution (Barrow, 1980). Consequently, as phosphorus is progressively added to soil the proportion of the added phosphorus in the adsorbed phase decreases while the proportion remaining in soil solution increases.

In the longer term (i.e. weeks to months) phosphorus initially adsorbed to the surface of oxides can migrate to surface sites between aggregates of crystals (Willett et al., 1988) and penetrate the crystal lattice (Barrow, 1989a). The phosphorus that is initially removed from solution by adsorption and continues to react with soil in these ‘slow’ reactions is commonly referred to as ‘fixed’ because it is no longer in direct equilibrium with the soil solution (Barrow and Shaw, 1975). Given the complexity of phosphorus reactions with soil and their time dependency, the terms ‘fixed’ and ‘fixation’ will be used to describe any inorganic reactions or processes that decrease the concentration of phosphorus in soil solution.

The phosphorus buffering capacity is a commonly used measure of a soil’s ability to maintain short-term inorganic phosphorus concentrations in the soil solution and, presumably, in other water such as overland flow. It is most often derived from a plot comparing the quantities of phosphorus adsorbed by soil to various equilibrium phosphorus concentrations under standard conditions. The slope of the graph at an arbitrary solution phosphorus concentration is referred to as the buffering capacity (Holford, 1989). An alternative test, also referred to as the phosphorus buffering capacity, uses the slope of a straight line for the relationship between adsorbed phosphorus and the log of the equilibrium phosphorus concentration (Rayment and Higginson, 1992). The higher the soil buffering capacity, the higher the proportion of phosphorus in the solid phase compared to the solution phase and the lower the rate of diffusion through the solution phase (Holford, 1989).

The buffering capacity is affected by the number of sites where phosphorus can be held, regardless of whether they are adsorption or precipitation sites, and the affinity of these sites for phosphorus (Holford, 1989). The decreasing slope of the plot comparing adsorbed phosphorus to various equilibrium phosphorus concentrations is consistent with a model in which the number of available sites and their affinity for phosphorus progressively decrease (Barrow, 1989a). It follows that as phosphorus is added to soil, for example through wastewater additions, the buffering capacity decreases. Further, since sorption plots are curvilinear, the higher the soil fertility the greater the decrease in buffering capacity from further fertiliser additions.

Buffering capacity is a useful way of comparing how soils respond to increased phosphorus in soil water. However, when phosphorus concentrations decrease, for example after rain, the dynamic processes involved in phosphorus adsorption, especially the longer-term processes such as intra-crystalline migration, are not simply reversed (White, 1980; Barrow, 1983). A soil’s ‘replenishment capacity’ (i.e. ability to replenish phosphorus concentrations in soil water) (Holford, 1989) depends on the buffering capacity of the soil and the amount of phosphorus that can be re-mobilised (i.e. labile phosphorus) (Schofield, 1955). As might be expected the ‘replenishment capacity’ is positively related to soil buffering capacity and the amount of labile phosphorus in soil. However, the export of phosphorus in flowing water will depend on both the replenishment capacity of the soil and the rate at which phosphorus can be mobilised.

The relationship between equilibrium phosphorus concentrations and inorganic phosphorus in various soil fractions has recently been the subject of renewed interest. A field investigation in England (Johnston, 1969) found a strong relationship between soil Olsen P (0-10 cm) and the phosphorus in sub-surface drainage from a single soil type (Heckrath, Brookes et al., 1995). A ‘change point’ was noted at Olsen P 57 mg P/kg, above which the change in phosphorus concentrations in drainage water per unit change in soil Olsen P increased dramatically. A number of studies have developed similar relationships (Maguire and Sims, 2002a, b), some measuring ‘change points’ as low as 10 mg P/kg Olsen P (Hesketh and Brookes, 2000).

The notion of a change point that is soil type specific is appealing from a regulatory point of view. However, it is more likely the ‘change point’ noted in these studies reflects the change between low- and high-energy adsorption sites (Barrow, 1983; Holford, 1997) to which a continuous curve would provide an equally, if not more, robust model for many soils.

From the point of view of a phosphorus source, the concept of a ‘change point’ is particularly relevant if it applied at or near the soil surface where most phosphorus is mobilised. However, there is little evidence suggesting that might be the case (Sharpley et al., 1978; Sharpley et al., 1982; Sharpley et al., 1985a; Sharpley, 1995). The relevance of a change point for phosphorus transported through sub-surface pathways depends on the interaction between the water and the soil after mobilisation.

Organic material can modify the adsorption/desorption properties of soil and as a consequence inorganic sources of phosphorus. Humic and fulvic acids compete for the sorption sites and as their concentrations increase, phosphorus adsorption decreases (Sibanda and Young, 1986). Organic ligands containing carboxyl acid groups appear particularly potent (Sibanda et al., 1986;

Holford, 1989) and, due to their variable charge, the severity of the organic matter interference with adsorption depends on pH (Barrow, 1989b). Other factors affecting direct organic matter interference with phosphorus adsorption include the absorbing surface composition and the nature of organic ions when in solution (Barrow, 1989b).

In addition to blocking adsorption sites, organic matter can also affect phosphorus adsorption indirectly by the chelation of iron and aluminium (Thomas, 1975; Bloom et al., 1979). Lowering the activity of these metals decreases the sites available for phosphorus adsorption, which in turn decreases the phosphorus adsorption capacity of soils high in organic matter (Holford, 1989; Tunney et al., 2000).

Organic matter clearly affects the ability of soil to supply inorganic phosphorus that can then be exported off-site (Weir, 1985; Holford, 1989; LeMare and Leon, 1990). However, the specificity of the reactions involved make it extremely difficult to interpret the relevance of these processes for wastewater irrigation sites other than to suspect that where organic matter in wastewater is added to soil, the availability of inorganic phosphorus would most likely be enhanced.

Organic phosphorus

Organic forms of phosphorus and their reactions in soil have received far less study than inorganic forms. Most organic phosphorus in soil originates from animal and plant wastes and the decomposition of soil flora and fauna. Faeces are the primary conduit through which animal phosphorus is returned to soil (Braithwaite, 1976). Cattle faeces are usually around 0.5% phosphorus on a dry weight basis (Thompson, 1989), while the figure for sheep faeces varies from 0.4-1.6% phosphorus depending on the time of year and soil fertility (Rowarth, 1987; Rowarth et al., 1988).

Strong evidence suggesting organic materials contribute directly to phosphorus exports can be found in the numerous studies in which organic and inorganic phosphorus concentrations in drainage have been compared (Reddy et al., 1978; Chardon and Oenema, 1995; Edwards et al., 1996; Chardon, Oenema et al., 1997). For example, phosphorus export in drainage from 300 mm deep soil cores treated with 50 kg P/ha as single superphosphate (mono-calcium phosphate plus gypsum) and cattle faeces have been compared (Nash and Murdoch, 1996b). In the case of the fertiliser, the phosphorus concentration in drainage was higher than the control cores only in the second week after treatment. However, the drainage from the faeces treated cores had significantly higher phosphorus concentrations than the control cores for the 5 weeks following treatment. The predominant forms of phosphorus in the drainage were also different, being dissolved reactive phosphorus (DRP, thought to be primarily orthophosphate) for fertiliser, and dissolved non-reactive phosphorus (DNP, presumably containing organic phosphorus) for the faeces treatments. The study concluded that while additions of phosphorus in fertiliser did not significantly increase the export of phosphorus in drainage, the application of phosphorus in faeces did. These results are consistent with long-term United Kingdom studies where the application of manures as compared to inorganic fertilisers applied at the same rate has increased phosphorus at depth in grass pastures (Johnston and Poulton, 1992).

It is unclear what enhances the mobility of phosphorus from organic matter such as faeces, compared to inorganic phosphorus sources. It may be the form of phosphorus in organic matter, a constituent of the organic matter, or a product derived from organisms using organic matter as a substrate (Dickinson and Craig, 1990).

Phosphorus mobilisation

Historically, the soil fabric has been assumed to act as a sink for applied phosphorus (Russell, 1957). Consequently, it was thought that phosphorus was primarily lost from soil when detached (eroded) sediments were exported in overland flow. The most notable exceptions were thought to be very sandy soils (Ozanne et al., 1961; Mansell et al., 1977; Peverill et al., 1977; Weaver, Ritchie et al., 1988a; Weaver et al., 1988b) and organic soils where soluble organic matter facilitated transport of phosphorus either directly or by coating the active sites for sorption (Pierzynski et al., 1994).

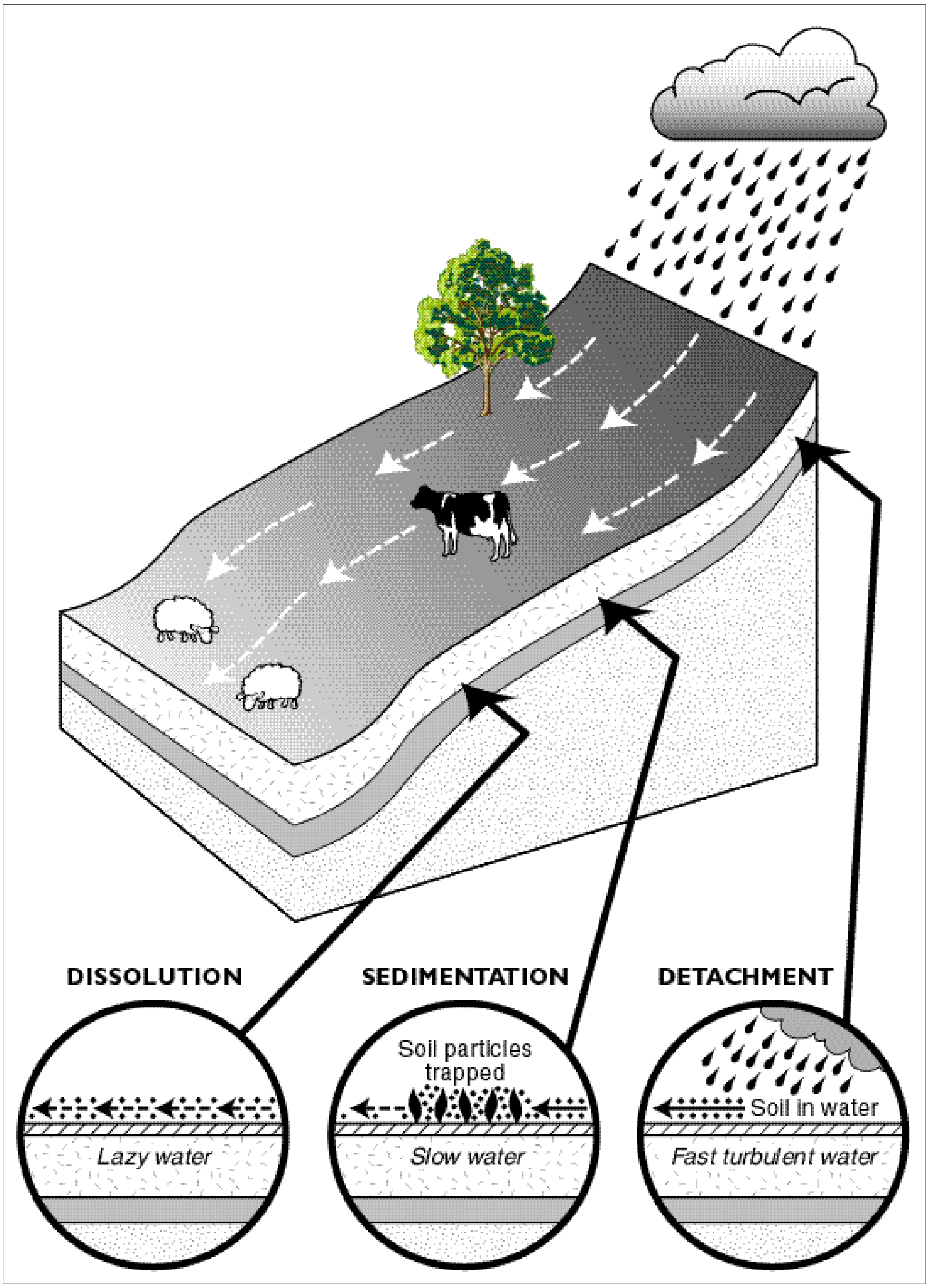
Filtration of water has generally been used to distinguish between detachment (erosion) processes that generate particulate phosphorus and dissolution processes that generate dissolved phosphorus. Typically, materials retained by a 0.45 μm filter have been defined as particulate and those in the filtrate as dissolved (Haygarth and Sharpley, 2000). The reactivity of phosphorus in an acid-molybdate solution is often used to further differentiate phosphorus species. Phosphorus concentrations measured without digestion are commonly referred to as ‘reactive’ and, with digestion, as ‘total’. The term ‘dissolution’ therefore defines the mobilisation of phosphorus and other materials that pass through a 0.45 μm filter but, importantly, the terms ‘dissolved’ and ‘dissolution’ do not imply that all the materials in the filtrate are in solution (Beckett and Hart, 1993; Haygarth et al., 1997). It is possible that wastewater reaction products and phosphorus attached to colloidal particles are contributing to phosphorus in the <0.45 μm or ‘dissolved’ fraction.

Both dissolved (<0.45 μm) and particulate phosphorus have been measured in overland flow from both grazing and cropping systems (Sharpley et al., 1981; Cullen, 1991; Hazel, 1991; Sharpley et al., 1991; Nash and Murdoch, 1996a; Nelson et al., 1996). This suggests that both physical detachment and dissolution processes (Figure 1) contribute to phosphorus exports. Soil has been shown to have a finite capacity to hold phosphorus and as this limit is approached the concentration of phosphorus in soil water increases (Barrow, 1989b; Holford, 1989; Heckrath et al., 1995; Hesketh et al., 2000). Where phosphorus in wastewater is applied to soil at higher rates than it is removed in produce, the amount of phosphorus stored in the soil will increase. It follows that the opportunities for dissolution of phosphorus into water are also increased at such sites.

While detachment (erosion) is essentially a physical process, dissolution is affected by the chemistry of the phosphorus and surrounding materials, and the available reaction time. Detachment commences with fine particles (sediments) and associated phosphorus being separated from the soil fabric by physical and mechanical effects on soil aggregates including raindrop impact, cultivation and flowing water, and physical and chemical effects on soil aggregate stability such as slaking and dispersion (Leeper and Uren, 1997). These particles are transported by overland flow at a rate that is related to the kinetic energy of the water (Shainberg et al., 1994). As a general rule, increased flow rates lead to increased potential for detachment and transport of sediment. This is likely on unstable soils where aggregates collapse, at higher rain intensities, on steeper slopes, on lower infiltration soils with fewer impediments to flow, and on slopes with significant run-on. The processes of surface soil detachment and sedimentation are well reviewed elsewhere (Smith and Wischmeier, 1962; Food and Agricultural Organisation of the United Nations, 1965; Kelley, 1983).

An important part of the detachment process is the sorting of particulate materials during transport based on their surface area and density. The disturbed flow created by perturbations at the soil surface enriches overland flow with phosphorus contained in low-density detrital material and phosphorus adsorbed to colloids, compared to the bulk source materials (Sharpley, 1980, 1985). Enrichment ratios (i.e. the concentration of phosphorus in the detached sediment divided by the concentration of phosphorus in the bulk source material) vary depending on the soil in question and soil treatment. As the concentration of detached soil increases, the enrichment ratio decreases and approaches unity (Massey and Jackson, 1952; Menzel, 1980). This suggests that phosphorus loads are higher, but concentrations lower, in more intense flows and has implications for water sampling strategies at the field-scale.

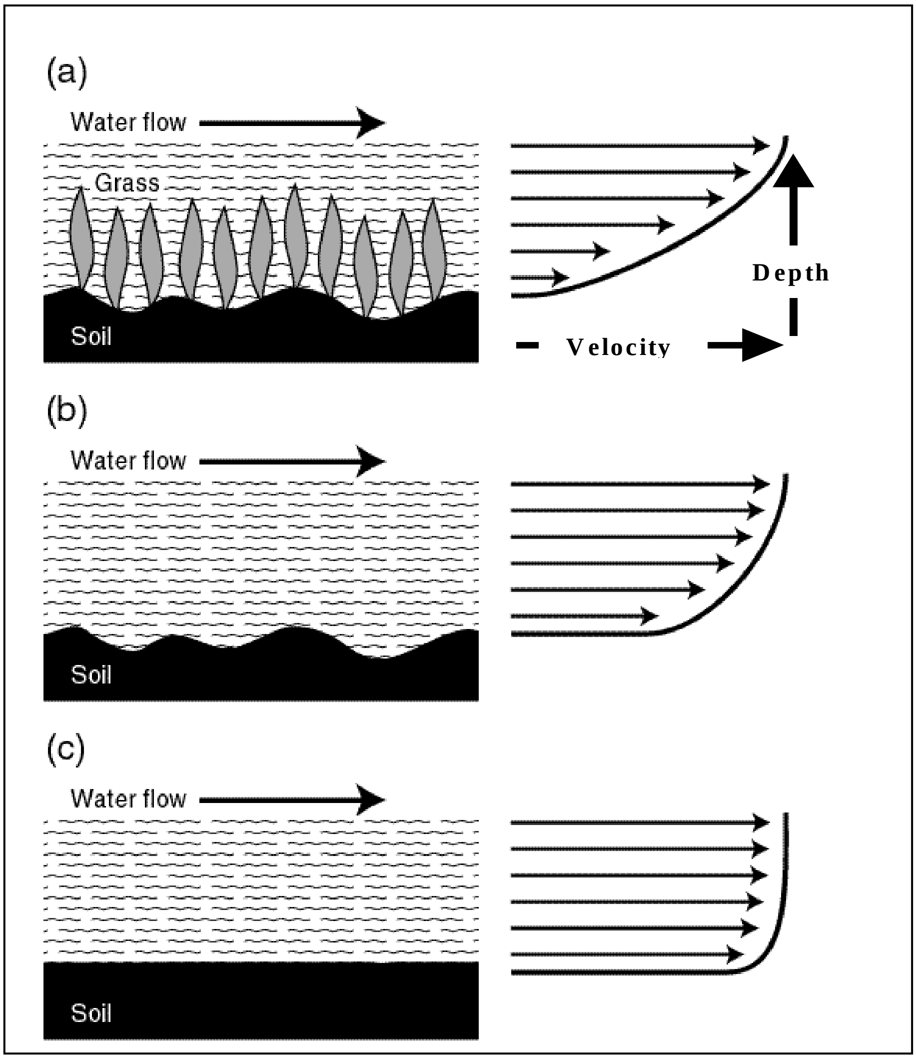
Figure 1. Phosphorus mobilisation at a field-scale.



Adapted from (Nash et al., 2002)

The sources of phosphorus mobilised by dissolution processes are generally close to the soil surface. However, unlike most pollutants, phosphorus can be extracted by water from both live and decomposing plant material (Bromfield et al., 1972; Sharpley, 1981). In contrast to detachment, dissolution occurs where the velocity of the water is low (Figures 1 and 2) and the migration of material into streamlines is determined by desorption, dissolution and diffusion rates rather than physical processes. A notable exception would be where raindrop impact on overland flow or detachment increases the surface area of a phosphorus source such as faecal materials (Ahuja et al., 1981; Ahuja et al., 1983).

Figure 2. The depth velocity profile of water moving down a slope with (a) vegetated surface, (b) rough soil surface and (c) smooth soil surface profiles.



Adapted from (Nash et al., 2002)

The most important factors affecting phosphorus dissolution rates include the volume of soil explored by the water (often only the first 20 mm layer), (Sharpley et al., 1988), the solubility of the phosphorus source, the sorption properties of the phosphorus species and soil, the presence or absence of chelating agents for metals or substances that block sorption sites, and the contact time (Sharpley et al., 1994; Kirkby et al., 1997; Addiscott and Thomas, 2000).

Where the depth of interaction between soil and water is essentially fixed, for example on stable soils, and source materials are not limiting, it would be expected that the load of dissolved phosphorus exported would depend almost exclusively on contact time. It follows that factors affecting flow rates such as the rain intensity, slope and depth of flow, have less effect on dissolved phosphorus loads than factors affecting the contact time between soil and water such as storm length and duration of flow (Haygarth and Jarvis, 1999; Haygarth et al., 2000). Using similar logic, and compared with detachment under the same conditions, dissolution is equally likely on upper and lower slopes.

Phosphorus transport

Surface pathways

The primary pathways for water-facilitated phosphorus transport in surface soil are shown in Figure 3. These pathways are rarely as well defined as the diagram suggests and in most landscapes infiltration excess overland flow, saturation excess overland flow and interflow are difficult to distinguish.

Infiltration excess overland flow occurs where the rainfall intensity and run-on exceed the infiltration rate for the soil profile. The volume and rate of infiltration excess overland flow depends on the rate of water addition, surface soil infiltration rate and soil hydraulic conductivity (Emmett, 1978). Consequently, factors such as slaking and dispersion that alter infiltration behaviour, and soil compaction and structural deterioration that affect hydraulic conductivity (Hillel, 1980; Cresswell et al., 1992), also affect infiltration excess overland flow. At a field-scale, infiltration excess overland flow tends to increase down slopes as both surface and sub-surface run-on from higher areas increase the hydraulic load, and hence the probability of overland flow, in receiving areas.

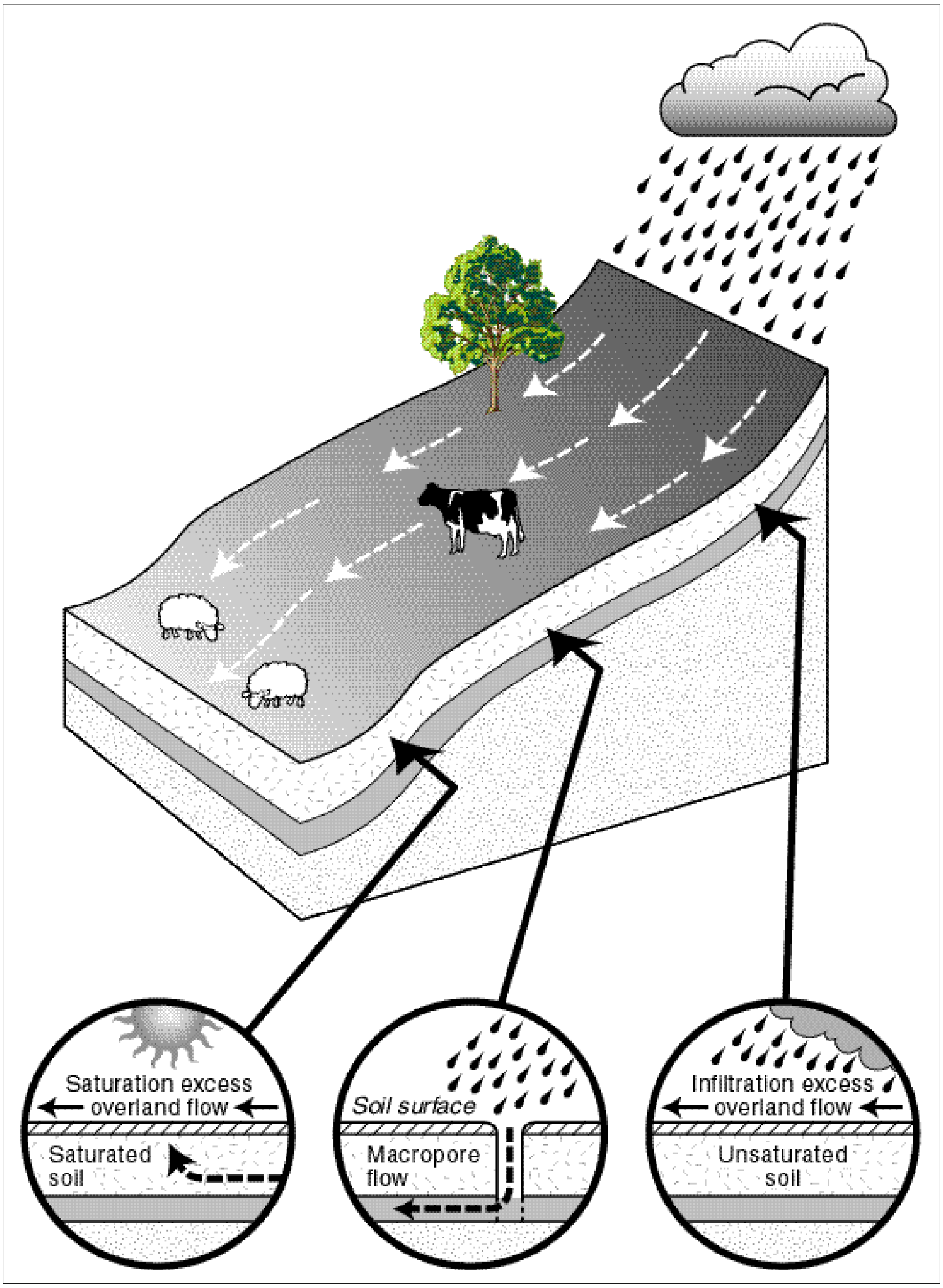
Saturation excess overland flow is characterised by saturation of the soil over which water is moving. For many soil profiles, saturation excess overland flow is a special case of infiltration excess overland flow. Generally, infiltration is occurring, albeit at a negligible rate because of the low hydraulic conductivity of the underlying strata. Soils with a permeable A-horizon overlying a heavy clay, less permeable B-horizon (Chittleborough, 1992) are prone to such conditions (Cox and McFarlane, 1995). However, saturation excess overland flow also occurs where ground water or interflow rises to the surface in discharge zones, often at a break (i.e. change) of slope (Rulon et al., 1985; Gerits et al., 1990; Moore and Foster, 1990).

Interflow describes water that after having infiltrated the soil is moving laterally until discharged as overland flow (Amerman, 1965), typically at the break of slope or where the profile has been disrupted (i.e. by clearing or ploughing). As such interflow can be considered a precursor of saturation excess overland flow. Also called ‘throughflow’ (Kirkby and Chorley, 1967), interflow can occur in the pollutant rich layer immediately below the soil surface or at the interface of the A/B or B/C horizons (Stevens et al., 1999). Waters moving at these interfaces are best referred to as A/B or B/C interflows and treated as components of sub-soil hydrology (Stevens et al., 1999).

As in the case of overland flow, interflow accumulates down a slope and when the infiltrating water and interflow exceed the transmissivity (i.e. hydraulic conductivity x soil thickness), infiltration/saturation excess overland flow is likely. Interestingly, interflow and infiltration excess overland flow can occur simultaneously, especially in naturally anisotropic (i.e. a soil profile having different physical properties in vertical and horizontal planes) soils or soils with a compaction layer near the surface. As infiltration/saturation excess overland flow and interflow immediately below the soil surface (0-20 mm) are difficult to separate functionally, the term overland flow will be used to describe them collectively.

Overland flow is one of the more difficult hydrological concepts to describe mathematically as water addition (rainfall) and infiltration vary temporally and spatially (Nielsen et al., 1973). In addition, flow may be laminar (subcritical), turbulent (supercritical) or a combination of both (Daugherty et al., 1989). Such characteristics affect mobilisation processes such as detachment.

Figure 3. Primary pathways of water and phosphorus transport at a field scale.



Adapted from (Nash et al., 2002)

Sub-surface pathways

The role of sub-surface pathways in the transport of phosphorus has received increasing attention in recent years (Smetten et al., 1994; Fleming and Cox, 1998; Stevens et al., 1999). On sloping land, water first infiltrates vertically and then moves laterally as interflow, often above a restricting layer such as a clay subsoil (Figure 1). As water moves through the soil, exchange takes place between water-borne constituents and the soil matrix. Soil material may be deposited or mobilised. As the accumulated interflow moves down slope it can express itself as interflow in another horizon or in an extreme case as saturation excess overland flow. Thus, water and phosphorus transported on the C horizon of 'leaky' texture-contrast soils in the upper parts of landscapes (Kirkby et al., 1996) may leave the catchment area as overland flow or flow on the B horizon (Stevens et al., 1999). The residence time in these pathways depends on a few parameters that are easily measured such as the flow gradient, path length and hydraulic conductivity and can therefore be readily calculated.

Where water moves slowly through the soil matrix (matrix flow) particulate phosphorus is removed and adsorption/desorption reactions decrease or increase dissolved phosphorus concentrations. However, where water moves quickly into and through soils via stable macropores (fissures or biopores), large quantities of water and contaminants can be rapidly transported down-slope (macropore flow). Many studies have demonstrated the ability of macropore flow to rapidly transport phosphorus through soil (Bottcher et al., 1981; Richard and Steenhuis, 1988; Van Ommen et al., 1989; Kladivko, Van Scoyoc et al., 1991), especially those with artificial drainage (Nash and Halliwell, 1999), when compared with matrix flow (Peverill and Douglas, 1976; Anderson and Bourma, 1977a, b; Kanchanasut et al., 1978; Seyfried and Rao, 1987; Singh and Kanwar, 1991; Small et al., 1994).

Originally, it was thought that to initiate macropore flow, the soil must be saturated, with water ponded over the entire soil surface (Beven and Germann, 1982). Later studies have shown that macropore flow could occur under non-ponded conditions provided saturated layers formed very near the soil surface (Adreini and Steenhuis, 1990) or at distinct horizon boundaries or structural discontinuities (Steenhuis et al., 1990). However, there is now growing evidence that macropore flow may occur without any such saturated zones. Methylene blue has been used to trace the pathways of water flow in unsaturated, undisturbed cores of a well structured, dry clay soil and this demonstrated that over half of the applied water left the cores via bypass or macropore flow (Booltink and Bourma, 1991). It has also been demonstrated that water from an unsaturated soil matrix can enter an artificially created macropore if the macropore walls are already wet (Philip, Knight et al., 1989). It would appear that, particularly in well aggregated soils, individual peds may become saturated or develop a film of water on their surfaces that can be channelled into nearby macropores without entering the unsaturated soil matrix (Bouma et al., 1979; Russell and Ewel, 1985; Seyfried et al., 1987; Sollins and Radulovich, 1988; Booltink et al., 1991; Trojan and Linden, 1992).

Discussion

It is possible to make a number of generalisations about export pathways. The physical size of most detached sediments prevents them moving through soil so particulate phosphorus is generally exported in overland flow. A notable exception is the phosphorus attached to fine colloids that move through macropores (Frenkel et al., 1978; Cox et al., 2000). Consequently, factors that increase the kinetic energy of overland flow near the soil surface such as high intensity rainfall, a lack of vegetation, high slopes and run-on, increase the export of particulate phosphorus.

Phosphorus mobilised by dissolution can be adsorbed onto soil during transit making it more difficult to generalise about its transport. Depending on the site at which mobilisation occurs, either at the soil surface or within the soil profile, and the relative amounts of water moving over or through the soil, either transport pathway may predominate. As an approximation, the level of soil and water interaction determines the relative importance of the pathways for dissolved phosphorus transport. Consequently, the highest concentrations of dissolved phosphorus are generally found in overland flow (Greenhill et al., 1983b; Greenhill et al., 1983c; Kirkby et al., 1996; Fleming et al., 1998; Stevens et al., 1999; Fleming, Cox et al., 2001). However, significant concentrations of the dissolved phosphorus can occur with macropore flow (Cox et al., 2000).

Unfortunately, while it is possible to generalise about the concentrations of phosphorus exported through various pathways, less is known about the overall loads. Being the product of concentration and flow volume, the loads of phosphorus exported through the different pathways depend on the hydrology of the field. For example, in many environments overland flow comprises a small percentage (<10%) of the total water applied. While the concentration of phosphorus in overland flow may be high compared with that in sub-surface water, the opportunity for the export of greater loads through sub-surface pathways is considerable (Stevens et al., 1999).

Mathematical modelling is a useful way of investigating and in some cases quantifying phosphorus mobilisation at the field scale (Trudgill 1995). Predictive tools that are used include regression equations, empirical relationships and simulation models (Sharpley et al., 1981a; Sharpley et al., 1982; Sharpley et al., 1985; Sharpley and Smith, 1989; Cooke et al., 1995; Hutson and Wagenet, 1995; Wagenet and Hutson, 1996; Nash and Murdoch, 1997; Nash et al, 2000). For investigating pollutant mobilisation comprehensive models that simulate soil, water and chemical processes are by far the most appealing. Unfortunately, as the processes they describe vary both spatially and temporally these simulation models are often extremely complex requiring vast data sets for calibration and the errors can be significant (Cambardella et al., 1994; Wagenet and Hutson, 1995).

Wastewater constituents that affect phosphorus mobilisation and transport

Depending on their source and pre-treatment, many wastewaters contain contaminants that can adversely affect soil/plant systems to which they are applied. In so doing these contaminants also affect phosphorus mobilisation and transport. In most cases these effects are expressed through changes in physical properties that result in anaerobic soil conditions. There are also risks associated with specific ion and other toxicities. These need to be assessed on a case by case basis and are well reviewed elsewhere (Ayers and Westcot, 1994; Maas, 1996) and will not reviewed in this report.

Salt is a component of wastewaters that affects both plant growth and soil physical properties. Irrigation of agricultural crops using with saline water and wastewaters has been extensively studied (Richards, 1954). Limitations on the use of saline water for irrigation include the accumulation of salts in the root zone and increased percentages of sodium adsorbed by clays.

Evapotranspiration of saline irrigation water increases salt concentrations within the root zone of crops. It follows that the ‘salinity hazard’ or risk to productivity increases with the salt concentration, and the use of saline irrigation water requires that additional water beyond the needs of the crop, a leaching fraction, is applied in order to remove accumulated salts. It follows that where drainage is restricted, for example in clay soils, the salinity hazard of irrigation water generally increases. This is reflected in the irrigation water classification systems that consider both the salt concentration of the water and the characteristics of the soil to which it is applied (Refer Table 2) (Hart, 1974; Ayers et al., 1994).

Table 2. General guidelines[a] for the salinity of irrigation water.

Water	EC (dS/m)	TDS (mg/L)	Application
Fresh	<0.7	<450	no restrictions on use, potable, all crops
Brackish	0.7-3.0	450-2000	slight to moderate restrictions on use, livestock water, most crops
Saline	>3.0	>2000	severe restrictions on use, salt-tolerant crops, not used on soils with restricted drainage

Adapted from (Ayers et al., 1994)
[a] EC, Electrical Conductivity.TDS, Total Dissolved Salts.

Salt in the root zone also affects soil structure. Compared to calcium and magnesium, sodium adsorption increases the tendency of clay aggregates to swell and disperse (van Olphen, 1977; Rengasamy and Olsson, 1991, 1993). The percentage of adsorbed sodium (ESP) is related to the Sodium Adsorption Ratio (SAR) of the soil solution (or applied water). At the ESP range most common in agricultural soils (ESP<30) the numerical values of ESP and SAR are almost equal (Shainberg and Letey, 1984). The ‘sodium hazard’ of irrigation water is therefore related to the SAR of irrigation water. It is of note that where salts are concentrated by evapotranspiration and as a result, the proportions of sodium, calcium and magnesium in solution are preserved, the SAR and sodicity hazard of that water increases. Consequently, some authors have proposed leaching requirements to assist with sodicity management (Rhoades, 1968).

$$SAR = [Na^+] / (0.5\{[Ca^{2+}]+[Mg^{2+}]\})^{0.5} \qquad \textit{Equation 1}$$

Where $[Na^+]$ = Sodium concentration (activity) of the solution in meq/l
$[Ca^{2+}]$ = Calcium concentration (activity) in the solution in meq/l
$[Mg^{2+}]$ = Magnesium concentration (activity) in the solution in meq/l

Even small changes in the ESP (i.e. <5%) can adversely affect soil physical properties (McIntyre, 1979). As the SAR of irrigation water increases, sodium is initially adsorbed to the surface of clay domains (i.e. quasi-crystals, groups of clay platelets) within aggregates (Sumner, 1993). Sodium adsorption increases the tendency for water to penetrate the space between the domains, increasing the distance between the domains and adversely affecting the ability of short-range attractive forces to hold the domains together. This potentially results in dispersive behaviour. As the ESP increases, sodium can penetrate the domains themselves increasing electrostatic repulsion between the platelets. As a result the clays swell and disperse. This phenomenon of initial sodium adsorption to clay domains within aggregates or 'demixing' is thought to account for dispersion being the dominant process at low ESP values (<15-25) and swelling at high ESP values (Halliwell et al., 2001).

The effects of a particular SAR/ESP depend on soil properties including textural class (i.e. clay percentage), clay type, quantity and type of entities that stabilise clay aggregates (i.e. organic matter) (Balks, Bond et al., 1998). But by far the most important factor affecting the expression of a particular ESP is the electrolyte concentration in the soil water. Electrostatic repulsion between moieties is greatly increased by lower electrolyte concentrations. Consequently, it is only when high quality irrigation water (i.e. low salt) is applied or rainfall occurs that the full effects of adsorbed sodium are expressed in the form of swelling and dispersion.

The effects of adsorbed sodium are particularly important in grassland and forage production systems. The physical stresses placed on the soil by animal and vehicular traffic, cultivation and the like can exacerbate adverse changes in soil structure, especially where soils are waterlogged. Increasing soil organic matter, for example as a result of organic matter being added to soil in wasetwater, can lessen such effects or indeed exacerbate them (Reid et al., 1982; Tisdall and Oades, 1982; Piccolo and Mbagwu, 1989; Tisdall, 1991; Nelson et al., 1999).

Many studies have reported decreases in soil infiltration rates and hydraulic conductivities, following wastewater irrigation (Day et al., 1972; Bouwer, 1973). Some of the mechanisms that could be responsible for these changes are outlined in Table 3. The swelling and dispersion of clay aggregates is likely to be particularly important as many wastewater constituents increase the risks associated with adsorbed sodium (Table 3) and soil ESP generally increases following wastewater irrigation (Kardos and Sopper, 1974; Kardos et al., 1974; Richenderfer et al., 1975; Murmann and Iskandar, 1977). Whether increased ESP results in changes in soil physical properties or are themselves the result of changes in soil properties, such as hydraulic conductivity, is an open question.

Table 3. Possible causes of changes in infiltration rates and hydraulic conductivities in soils following wastewater irrigation.

Mechanism	Source
Solvent-solute effects on clays.	(Michel et al., 2000; Anandarajah, 2003)
Accumulation of suspended solids at the soil surface or blockage of the inter-soil spaces by suspended material such as colloidal clay and cells from microorganisms.	(Berend, 1967; Laak, 1970; De Vries, 1972; Bouwer and Chaney, 1974; Metzger, Yaron et al., 1983; Siegrist, 1987; Vandevivere and Baveye, 1992)
Entrapped air.	(Rice, 1974)
Formation of a biological mat or crust including the production of microbial extracellular polymeric materials such as polysaccharides.	(Thomas et al., 1966; Kristiansen, 1981; McAuliffe et al., 1982; Kawanishi et al., 1990; Taylor et al., 1990; Balks et al., 1997)
Collapse of soil structure due to organic matter dissolution.	(Lieffering and McLay, 1996)
Physico-chemical changes to pore geometry and micro-fabric that are related to sodicity, cation exchange reactions and exchange hysterisis.	(Shainberg et al., 1971; Shainberg et al., 1984; Verburg and Baveye, 1995; Yong, 1999)

Adapted from (Halliwell et al., 2001)

The interactions between soil and wastewater are extremely complex and it would appear that a number of wastewater constituents might act in concert, increasing the risk that irrigation systems fail. Defining the mechanisms by which wastewater constituents affect soil properties warrants further investigation in the context of a system where the physical (i.e. soil type and characteristics, infrastructure), environmental (i.e. climate) and cultural/social factors are considered. Consider, for example, irrigation using poorly treated abattoir wastewater in a semi-arid tropical environment. Suspended sediment may initially restrict soil infiltration rates. Sodium adsorption increases causing changes to the clay fabric. Rainfall lowers electrolyte concentrations during the ‘wet’ season and physically disrupts aggregates at the soil surface. Dispersion products further restrict drainage. Salt concentrations in the root zone increase during the irrigation season resulting in further increases in the ESP. The swelling of clay aggregates decreases porosity and oxygen diffusion is suppressed. Stimulated by soluble (labile) organics, anaerobic soil conditions predominate for part of the year. Some key cations, such as iron (III), are reduced and solubilised, resulting in further destabilisation of aggregates. Dispersion products further block soil pores, lowering hydraulic conductivity. Leaching rates decline further until eventually the leaching required to maintain an adequate salt balance in the root zone cannot be sustained and the system fails.

Table 4. Constituents that increase the effective SAR and/or sodicity hazard of wastewaters

Constituent	Possible Mechanisms	Source
Inorganic anions, especially SO_4^{2-}, CO_3^{2-}, HCO_3^-.	Precipitation of Ca^{2+} and Mg^{2+} from solution. Only occurs at high electrolyte concentrations	(Bower et al., 1968; Paliwal and Deo, 1978; Suarez, 1981; Ayers et al., 1994)
Organic matter	Chelation of Ca^{2+} and Mg^{2+}; Chelation of cations that would otherwise form bridges between minerals; Altering the conformation of organic molecules favouring their association with a single mineral rather than between particles; Anaerobic or other conditions resulting in the chemical reduction or lowered activity of cations that would otherwise form bridges between minerals (i.e. Fe^{3+}).	(Pistol, 1981; Reid et al., 1982; Metzger et al., 1983; Shainberg et al., 1984; Visser and Caillier, 1988; Piccolo et al., 1989; Balks et al., 1998)

Concluding comments

Plants take up phosphorus from soil water. It follows that phosphorus exports in surface and sub-surface drainage are a natural consequence of any plant production system. Wastewater irrigation sites also export phosphorus. The important question is do they export more phosphorus than an equivalent site where freshwater is used for irrigation.

Most phosphorus exports are initiated at or near the soil surface. It is relatively easy to list the different forms of phosphorus and their reactivity. However, the complexity of biological systems makes it extremely difficult to generalise about the importance of the various phosphorus sources.

Phosphorus mobilisation processes are also easy to describe. The physical detachment and transport (erosion) of soil particles carrying phosphorus have been extensively studied and remedial strategies developed. There is no reason to believe that mobilisation of sediment should be important at well-managed wastewater irrigation sites. Dissolution is a far more complex process and less easily managed. Dissolution of phosphorus depends on the phosphorus sources and a range of soil physical and chemical properties that often change as a result of wastewater irrigation.

As a general rule phosphorus mobilised by detachment processes is transported through surface pathways. Dissolution products on the other hand can be transported through both surface and sub-surface pathways. Due to the adsorption of phosphorus in sub-soil, the highest concentrations of phosphorus are generally found in surface waters. However, the load of phosphorus exported from any particular site is a function of both concentration and flow. It follows that environmentally significant quantities of phosphorus may also be transported through sub-surface pathways.

It is not possible to say if wastewater irrigation sites export more phosphorus than an equivalent freshwater irrigation site. The answer will depend on, amongst other things, the relative fertility of the sites, soil macroporosity, the effects of wastewater on the soil physical and chemical properties, and most importantly, the management of the respective sites.

Regulatory approaches to the application to land of abattoir wastes containing phosphorus in selected countries

Introduction

The regulation of abattoir wastewater applications to land is currently under review in a number of countries. In many cases, nitrogen rather than phosphorus has been used for determining the rates at which abattoir wastes are applied to land. However, there is an increasing trend towards utilising wastes as sources of nutrients for agricultural production rather than disposing of them on land. Phosphorus has a stronger tendency to accumulate in soil than nitrogen. As a result, changing the emphasis from disposal to utilisation is likely to increase the importance of phosphorus in determining waste application rates and the longevity of waste application sites.

United Kingdom

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Waste applications to land in the United Kingdom are managed under the auspices of the Waste Framework Directive 75/442/EEC and an amending directive 91/156/EEC. The implementation of these into United Kingdom legislation is done through the Waste Management Licensing Regulations 1994 (WMLR) and its amendments. Devolution may mean there are slight differences in Scotland and Ireland and possibly Wales.

The United Kingdom legislation does not set phosphorus limits for waste application to land. Blood and gut contents from abattoirs, and presumably some associated water, are currently spread on land by way of an exemption from the need for a license under the WMLR. The exemption does not specify phosphorus limits for the application to land of blood and gut contents or any other exempted wastes.

In the United Kingdom there are limits on the quantities of nitrogen that can be applied to land. However, while there are no legislative limits on phosphorus applications, good practice (ie voluntary) advice is provided. That advice is primarily that where soil phosphorus as measured by soil tests is agronomically adequate or above, the total inputs of phosphorus from all sources should not exceed that removed in produce.

The government is going to amend the WMLR's and published a consultation paper on June 27, 2003. In part the intention is to toughen up the exemptions such that the onus is on the waste producer (or at least whoever applies it to land, since that is the activity that is exempted) to prove that there is an agricultural benefit to applying wastes to land to the satisfaction of the regulator (in the United Kingdom the Environment Agency). This has been the result of some abuses of the exemption process.

The Department of Environment, Food and Rural Affairs is still dealing with the issue of waste applications to land and no amending regulations have been laid as yet. However, as the intent of the probable changes will be to place more emphasis on the use of wastes as a source of nutrients for agronomic production rather than the disposal of wastes on land, it is almost certain that in future, land application of wastes will take account of all nutrients including phosphorus.

The Netherlands

Information sources:

Dutch Meat Board
www.hollandmeat.nl

VROM International
Netherlands Ministry of Spatial Planning and the Environment
www2.vron.nl

Ministry of Agriculture, Nature Management and Fisheries
Infoteik
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The Dutch government first introduced legislation to address the issue of mineral surpluses in agricultural systems in the 1980's. These regulations have been progressively tightened. However, while this is a holistic approach to nutrient management, as in the United Kingdom initial emphasis seems to have been on nitrogen rather than phosphorus. Indeed at the current minerals management system “Minas”, is currently being adjusted to comply with the EU Nitrates Directive (91/676/EEC) after the European court of Justice found that elements of the Netherlands fertiliser policy are not in line with European requirements. It is of note that the Netherlands has agreed with the European Commission that a nutrient balance will be achieved for phosphate by 2015.

Minas is an input-output budgeting system that sets restrictively high levies on phosphate and nitrogen surpluses above a certain maximum allowed for nitrogen and phosphorus per hectare known as the “loss standards”. Farmers keep detailed records of nitrogen and phosphorus inputs and outputs on their farm and are required by law to complete annual mineral returns and submit them to a central registry. This effectively forces farmers to take measures to minimise mineral losses to the environment. Since January 1, 2002 a manure trading system has enabled farmers who produce excess manure, and presumably wastes such as those from animal processing, to dispose of their surpluses by entering into agreements with arable farmers and manure/waste processors. However, it remains unclear if abattoir wastewater irrigation sites are subject to Minas.

The loss standards for phosphate are presented in Table 5. These standards are relatively generous by Australian standards. However, it is of note that anecdotal evidence suggests that the loss standards have been modified since this publication was produced (Hugo van der Meer, November 2004).

Table 5 The “loss standards” used in the Minas system for determining phosphorus surpluses.

Year	Phosphorus loss standard[a] (kg/ha annually)	
	Arable land	Grassland
2001	35	35
2002	30	25
2003>	20	20

(Source: Department of Agriculture, 2001)

United States

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A number of scientists based in the United States were contacted for information. It would appear that abattoirs in the United States are currently licensed on a case by case basis but nitrogen rather than phosphorus is often used for calculating loading rates. Phosphorus Indices have been developed in order to regulate phosphorus applications to land in much of the country. However, both Andrew Sharpley and Phillip Moore have been involved in the development of phosphorus indices and neither scientist was aware of the indices being applied to abattoir wastewater irrigation sites. There are no groundwater phosphorus limits in the United States.

Australia

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In Australia there appears to be general guidelines for the application of wastewaters to land but most land application systems for abattoir wastewaters are licensed on a case by case basis. The guiding principles are that there should be no surface runoff from irrigation, surface and groundwater quality should be protected and there should be no harm to the environment (including the soil). Whether a site needs a license varies from state to state, with 3000 kg of product per day being the trigger for licensing premises, and therefore wastewater irrigation systems, in New South Wales. Soil tests are used to assess risk factors associated with phosphorus accumulation at most sites.

In Victoria the phosphorus in abattoir wastewaters applied to land is usually licensed. However, the concentrations of phosphorus specified in licenses, for example 20 mg P/L and 5 mg P/L for two abattoirs in Gippsland, can be unrealistically low given that most wastewaters receive only primary screening and lagoon treatment is aimed at lowering the BOD_5 rather than phosphorus concentrations. As a consequence some abattoirs are regularly in breach of their license conditions.

In Victoria, the guidelines for application of wastewater phosphorus to land recommend that annual applications of phosphorus in wastewater be limited to 80 kg P/ha annually (EPA, 1996). With many abattoir wastewaters having up to 30 mg P/L, the 80 kg P/ha recommendation is often exceeded, especially in Northern Victoria where wastewater irrigation rates can be in the order of 8-9 ML annually. Site monitoring in the form of soil testing is used to assess the build-up of nutrients over time.

In New South Wales the situation is similar to Victoria with regulations that are framed in terms of outcomes (i.e. no water pollution and no harm to the environment) and guidelines as to how this can be best achieved (new Effluent Re-use Guidelines are about to be released). Some licenses contain limits on phosphorus concentrations and volumes of wastewater that may be applied to land, providing an upper limit on soil phosphorus loadings.

While there are no specific limits to phosphorus applications in South Australia, increasing emphasis is being placed on the development of irrigation management plans that incorporate all aspects of the system under consideration. The ultimate aim would be, where possible, to balance nutrient inputs and outputs. With this in mind the Environment Protection Authority commissioned the development of a spreadsheet to assist in calculating wastewater application rates.

The South Australian “Manual for Spreading Nutrient Rich Wastes on Agricultural Land” (Clarke, 2003) is basically a monitoring protocol (Trevor Clarke, pers. comm., November 29 2004). Data is input to the spreadsheet and when the equilibrium phosphorus concentration measured using a Phosphorus Retention Index (22°, 0.02 M KCl containing 10 mg P/L, 1:20 soil-solution ratio) is greater than 0.5 mg P/L for the root zone (including the subsoil), it is assumed that the phosphorus adsorption capacity of the soil is exceeded, and any further inputs of phosphorus must equal outputs. This sets an upper limit on phosphorus additions that is based on soil testing and as such allows for time-dependent fixation reactions. Whether the 0.5 mg P/L Phosphorus Retention Index is appropriate is an open question as it was based on feedlot studies from Queensland (Trevor Clarke, pers. comm., November 29 2004).

Western Australia also regulates abattoir wastewater application sites on a case by case basis and the license criteria are rarely based on phosphorus. Heavy metal contamination is considered a very important issue for wastewater irrigation systems. The Western Australian Government is currently adapting the South Australian spreadsheet for local conditions. It could be expected that, if and when the spreadsheet is introduced, there will be greater emphasis on nutrient management at wastewater irrigation sites.

In Queensland the application of wastewaters to land is governed by a management plan that considers a range of contaminants. Soil test values are used to monitor the sites for phosphorus.

Tools for regulating the application of meat processing wastewaters to land

Soil testing

Soil testing is one of the tools used by regulatory authorities to monitor wastewater irrigation sites and assess their potential offsite impact. Many of these tests use a chemical extractant to estimate the size of various "pools" (i.e. sources) of phosphorus as determined by the particular soil sampling and extraction procedure. Standard agronomic soil tests, such as Olsen P or Colwell P that are commonly used to estimate soil fertility in the root zone (i.e. 0-10 or 0-15 cm) fall into this category. Such tests are generally distinguished on the basis of whether they estimate the "quantity" of soil phosphorus that is available for uptake during the crop cycle or the "intensity" (i.e. activity) of phosphorus in soil solution (Moody and Bolland, 1999). Both the Olsen P and Colwell P tests use a bicarbonate solution (0.5 M $NaHCO_3$ at pH 8.5) to extract phosphorus. The shorter duration (30 min.) Olsen P test estimates both phosphorus quantity (i.e. supply) and intensity (i.e. concentration) while the Colwell P (16 hr.) predominantly measures phosphorus quantity (Moody et al., 1999). For environmental testing there is an increasing trend towards the use of short extraction periods (i.e. 0.25 to 1 hr.) and distilled water or dilute electrolyte (i.e. 0.01M $CaCl_2$) extractants in order to better predict phosphorus activity. Importantly, as result of the procedures used to analyse for phosphorus in the extractant, most commonly phosphomolybdenum blue chemistry (Murphy and Riley, 1962) or inductively coupled plasma atomic emission spectroscopy (ICP-AES), these tests measure classes of phosphorus compounds rather than the concentrations of individual species and only in the soil layer tested.

A second category of soil tests used for both agronomic and environmental purposes measure the capacity of soil to adsorb or desorb phosphorus. These tests measure phosphorus adsorption/desorption (mg P/kg soil) following the equilibration of soil in one or more solutions of differing phosphorus concentrations under standard conditions (Bache and Williams, 1971; Blakemore et al., 1981; Allen and Jeffery, 1990; Dear et al., 1992; Rayment et al., 1992; Barrow, 2000; Burkitt et al., 2002). The results of these tests are usually analysed either by fitting the Freundlich, single or double surface Langmuir or other equations, by fitting a regression model or by combining the results with other data, such as Olsen P or Colwell P, to develop various adsorption indices.

A third category of soil tests that are increasingly used for environmental purposes, particularly in the United States, relates the concentration of phosphorus extracted from a soil to other chemical species, most commonly iron and aluminium (called the degree of phosphorus saturation), in the same solution (Rayment et al., 1992), or to other tests (Pautler and Sims, 2000). The most common of these tests uses acid ammonium oxalate as the extractant. However, the use of oxalate as an extractant has been restricted to acid soils where amorphous iron and aluminium dominate phosphorus adsorption reactions rather than calcarious soils where calcium dominates phosphorus adsorption (Kleinman and Sharpley, 2002). As a consequence there has been increasing interest in developing similar indices using Mehlich extracts (Beck et al., 2004). Where the degree of phosphorus saturation is <20 to 25% there appears to be little potential for phosphorus export through leaching (Maguire et al., 2002b).

There have been many studies that have compared the different soil tests (Pautler et al., 2000; Kleinman et al., 2002; Beck et al., 2004) and their relationship to phosphorus mobilisation and transport in simulation studies (Pote et al., 1996; Pote, Daniel et al., 1999; Sharpley et al., 2001; Maguire et al., 2002a, b; Sims et al., 2002; Torbert, Daniel et al., 2002; Schroeder et al., 2004; Turner et al., 2004). Unfortunately, despite considerable investment in development and calibration, soil phosphorus tests remain an imprecise instrument. Soil phosphorus test results vary with the time of sampling and sampling protocol (Department of Natural Resources and Environment, 2001), sample preparation (i.e. drying) and the choice of laboratory (Rayment, 2004).

Despite their deficiencies, soil phosphorus tests are an invaluable tool for estimating the source component of phosphorus exports. However, the value of any soil test in predicting phosphorus export potential is limited to the system under investigation, soil depth at which the test is conducted and the primary pathway for phosphorus export. For example, the English study (Heckrath et al., 1995) in which a strong relationship was found between Olsen P (0-10 cm) and the phosphorus in drainage was conducted on a soil with tile drains. The authors noted that "…preferential flow or rapid transport of P in forms less susceptible to sorption but measured as DRP…" may account for the results. Therefore, it is not surprising that there was a relationship between soil test phosphorus in the area where mobilisation took place (i.e. at the soil surface) and the phosphorus in drainage. However, it is likely that no such a relationship would exist where phosphorus was transported in matrix flow through a low phosphorus sub-soil. Other authors have also noted the importance of the transport pathway (i.e. site hydrology) in determining phosphorus exports in surface pathways (Pote et al., 1999).

Phosphorus indices

A major criticism of using only soil test parameters for regulatory purposes is that they primarily reflect the source component of the system and, depending on the test, to a lesser extent mobilisation processes. Phosphorus Indices that combine quantitative and qualitative information relating to phosphorus sources, mobilisation and transport into a single number have been developed to rectify this deficiency (Lemunyon and Gilbert, 1993), albeit, to the authors knowledge, not for wastewater irrigation. Phosphorus Indices are a widely used in the United States (Table 6) to manage the application of phosphorus to agricultural land.

The Pennsylvania Index was one of the first developed and is similar to most others. To use the index, source factors are measured or rated including Soil Test Phosphorus, Fertiliser Rate, Fertiliser Application Method, Manure Application Rate, Manure Application Method and Manure Phosphorus Availability. A Soil Test Rating, Fertiliser Rating and Manure Rating are then calculated and summed to derive a Source Factor. A Transport Factor is calculated based on the sum of factors describing Erosion, Runoff Potential, Sub-surface Drainage and Contributing Distance, modified for Connectivity (i.e. riparian vegetation etc.). The Phosphorus Index Value is computed as twice the product of the Source Factor and the Transport Factor (Weld, Beegle et al., 2003). Combining the Source Factor and Transport Factor in this way allows for synergistic effects and acknowledges that either a high source potential or high transport potential, by themselves, are unlikely to lead to excessive phosphorus exports.

Phosphorus Indices are a simplistic representation of a complex biological system. They do not attempt to quantify phosphorus exports but rather give general guidance for management and the changes that are necessary protect water quality. The structures of the indices lend themselves to wastewater irrigation. Similar principles have recently been incorporated into the catchment scale, GIS based "Phosphorus Indicators Tool" developed in Europe (Heathwaite et al., 2003).

Table 6. Phosphorus Indices used in the United States.

State	Phosphorus Index Reference
Alabama	U.S. Department of Agriculture-Natural Resource Conversation Service. 2001. Phosphorus index for Alabama: A planning tool to assess and manage P movement USDA-NRCS Agronomy Technical Note AL-72. Auburn, Alabama USA.
Alaska	U.S. Department of Agriculture-Natural Resource Conversation Service. 2001. The Alaska phosphorus index. USDA-NRCS Technical Note Agronomy-14. Palmer, Alaska USA.
Arizona	Walther, D., R. Flynn, M. Sporcic, and L. Scheffe. 2000. Draft phosphorus assessment tool for Arizona. USDA-NRCS Technical Note 57. Phoenix, Arizona
Arkansas	DeLaune, P.B., P.A. Moore Jr., D.E. Carmen, T.C. Daniel, and A.N. Sharpley. 2001. Phosphorus Index for Pastures. University of Arkansas, Fayetteville, A.R.
Colorado	Sharkoff, J.L., R.M. Waskom, and J.G. Davis. 2000. Colorado phosphorus index risk assessment. USDA-NRCS Agronomy Technical Note CO-95. Lakewod, Colorado USA. (Available on-line at http://www.co.nrcs.usda.gov/ecs/technotes/coatn_95.pdf; verified 3 September 2002).
Delaware	Sims, J.T. and A.B. Leytem. 2002. The Phosphorus Index: a phosphorus management strategy for Delaware's agricultural soils. Fact Sheet ST-05, Delaware Cooperative Extension Service, Newark. DE.
Florida	U.S. Department of Agriculture-Natural Resource Conservation Service. 2000. The Florida phosphorus index. Florida Agronomy Field handbook, Chapter 1, Exhibit 9. Gainsville, Florida USA. (Available on-line at http://www.fl.nrcs.usda.gov/flgeneral/pifinal.pdf; verified 30 August 2002).
Georgia	Cabrera, M.L., D.H. Franklin, G.H. Harris, V.H. Jones, H.A. Kuykendall, D.E. Radcliffe, L.M. Rise, and C.C. Truman. 2002. The Georgia phosphorus index. Cooperative Extension Service, Publications Distribution Center, University of Georgia, Athens, Georgia, 4 pp.
Illinois	USDA-NRCS. 2002. Illnois phosphorus index procedure: Use and interpretation of the Illinois phosphorus assessment procedure. USDA-NRCS conservation practice standard: Nutrient management code 590. Champain, Illinois.
Iowa	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Iowa phosphorus index. USDA-NRCS Technical Note 25. Des Moines, Iowa USA (Available on-line at http://www.ia.nrcs.usda.gov/Technical/Phosphorus/phosphorusstandard.htm; verified 30 August 2002).
Kansas	Davis, B., G.M. Pierzynski, F.Vocasek, L. Freese, and G. Keeler. 1999. Nutrient management in Kansas. *In* SWCS abstracts, St. Louis, MO. 8-12 July 2000. SWCS, Ankeny, Iowa. (Available on-line at: http://www.swcs.org/t_publicaffairs_nutmgmt_kansas.htm; verified 6 November 2002).
Kentucky	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Kentucky phosphorus (P) matrix. USDA-NRCS Conservation practice standard: Nutrient management code 590. Lexington, Kentucky USA.
Louisiana	U.S. Department of Agriculture-Natural Resource Conservation Service. 2000. Phosphorus site index for Louisiana. USDA-NRCS Conservation practice standard: Nutrient management code 590. Alexandria, Louisiana USA.
Maine	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Environmental assessment and manure allocation tool. USDA-NRCS Conservation practice standard: Nutrient management code 590. Bangor, Maine USA.
Maryland	Coale, F. 2000. The Maryland phosphorus site index: An overview. Soil Fertility Management SFM-6. University of Maryland, College Park, MD USA (Available on-line at http://www.agnr.umd.edu/MCE/Publications/Publications.cfm?ID=537; verified 30 August 2002).
Michigan	Grigar, J., D. Pahl, and R.D von Bernuth. 2002. Instructions for using MARI on Excel (Available on-line at http://www.maeap.org/resources.htm#3; verified 30 October 2002).
Minnesota	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Phosphorus loss potential and manure application rates. USDA-NRCS Conservation practice standard: Nutrient management code 590. St. Paul, Minnesota USA.
Mississippi	U.S. Department of Agriculture-Natural Resource Conservation Service. 2000. Phosphorus index rating. USDA-NRCS Conservation practice standard: Nutrient management code 590. Jackson, Mississippi USA.
Montana	Fasching, A. 2001. Phosphorus index assessment for Montana. USDA-NRCS Technical Note Ecological Sciences Agronomy MT-77. Bozeman, Montana USA.
Nebraska	Kucera, M. 2000. Assessing and managing phosphorus loss for manure management. USDA-NRCS Agronomy Technical Note Draft. Lincoln, Nebraska USA.

New Hampshire	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Phosphorus index calculation sheet. USDA-NRCS Conservation practice standard: Nutrient management code 590. Durham, New Hampshire USA.
New Jersey	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. New Jersey phosphorus index for typical crop production systems. USDA-NRCS Conservation practice standard: Nutrient management code 590. Somerset, New Jersey USA.
New Mexico	Flynn, R., M. Sporcic, and L. Scheffe. 2000. Draft Phosphorus Assessment Tool. USDA-NRCS Technical Note Agronomy-57. Albuquerque, New Mexico USA. (Available on-line at http://www.nm.nrcs.usda.gov/techserv/TechNotes/agro.htm; verified 12 July 2002).
New York	Czymmek, K.J., Q.M. Ketterings, and L. Geohring. 2001. Phosphorus and Agriculture VII: The new phosphorus index for New York State. What's Cropping Up? 11 (4): 1-3. (Available on-line at http://www.css.cornell.edu/nmsp/projects/pindex.asp; verified 30 August 2002).
North Carolina	Havlin, J., S. Hodges, D. Osmond, A. Johnson, D. Crouse, W. Skaggs, R. Evans, J. Parsons, P. Westerman, L. Price, and R. Reich. 2002. Assessing the risk of Phosphorus delivery to North Carolina waters. *In* Proc. of ASAE Annual International Meeting, Chicago, IL. 28-31 July 2002. ASAE, St. Joseph, MI (Available on-line at http://www3.bae.ncsu.edu/SW21/P-session64/P-session-64-asae2002.pdf; verified 6 November 2002).
North Dakota	U.S. Department of Agriculture-Natural Resource Conservation Service. 2002. North Dakota phosphorus index screening tool. USDA-NRCS Technical Guide Notice ND-9. Bismarck, North Dakota USA.
Ohio	U.S. Department of Agriculture-Natural Resource Conservation Service. 2002. Phosphorus index risk assessment procedure worksheet. USDA-NRCS Ohio Field Office Technical Guide Section 1. Columbus, Ohio USA.
Oklahoma	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. Oklahoma phosphorus assessment worksheet. USDA-NRCS Conservation practice standard: Nutrient management code 590. Stillwater Oklahoma USA.
Oregon	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. The phosphorus index. USDA-NRCS Agronomy Technical Note 26. Portland, Oregon USA. (Available on-line at ftp://ftp.or.nrcs.usda.gov/pub/agronomy/Phosphorus_index/; verified 30 October 2002).
Pennsylvania	Weld, J.L., D.B. Beegle, W.J. Gburek, P.J.A. Kleinman, and A.N. Sharpley. 2003. The Pennsylvania phosphorus index: Version 1. Publications Distribution Center, Pennsylvania State University, University Park, PA. (Available on-line at panutrientmgmt.cas.psu.edu; verified 21 December 2003).
Rhode Island	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001k. Rhode Island phosphorus index (RIPI). USDA-NRCS Rhode Island Field Office Technical Guide Section II. Warick, RI USA.
South Carolina	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001l. The phosphorus index: South Carolina. Agricultural waste management field handbook, Chapter 11, South Carolina Supplement 2. Columbia, South Carolina USA.
Tennessee	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001m. Tennessee phosphorus index: A planning tool to assess and manage P movement. USDA-NRCS Nashville, Tennessee USA.
Texas	U.S. Department of Agriculture-Natural Resource Conservation Service. 2000. Phosphorus assessment tool for Texas. USDA-NRCS Agronomy Technical Note Number -15. Temple, Texas USA.
Utah	Goodrich, K.I., R.T. Koenig, S.D. Nelson, L.L. Young, N.P. Hansen, and J.W. Hardman. 2000. A procedure for determining best management practices for spreading of manure on agricultural land in Utah, the Utah manure application risk index (UMARI). USDA-NRCS Salt lake City, Utah
Vermont	Jokela, W.E. 2000. A phosphorus index for Vermont. P. 302-315. *In* Proc. From Managing nutrients and pathogens from agriculture. Camp Hill, P.A. 28-30 Mar., 2000. NRAES-130. Ithica, NY.
Virginia	Mullins, G., M.L. Wolfe, J. Pease, L. Zelazny, L. Daniels, M. Beck, M. Brosius, A. Vincent, and D. Johns. 2002. Virginia Phosphorus Index version 1technical guide. Virginia Polytechnic Institute and State University, Blacksburg, VA USA.
Washington	U.S. Department of Agriculture-Natural Resource Conservation Service. 2001. The phosphorus index. USDA-NRCS Water Quality Technical Note 2. Spokane, Washington USA.
West Virginia	U.S. Department of Agriculture-Natural Resource Conservation Service. 2002. Phosphorus index for nutrient management. USDA-NRCS Conservation practice standard: Nutrient management code 590. Morgantown, West Virginia USA.
Wisconsin	Jarrell, W. and L. Bundy. 2002. The Wisconsin phosphorus index. University of Wisconsin Madison, WI USA. (Available on-line at: http://wpindex.soils.wisc.edu/index.html; verified 4 November 2002).

Wyoming	U.S. Department of Agriculture-Natural Resource Conservation Service. 2002. The phosphorus index. Section 12- Nutrient basis for manure application rates Wyoming comprehensive nutrient management plan workbook. Casper, Wyoming USA.

Numerical models

Numerical models are an increasingly popular way of quantitatively assessing the environmental impacts of pollutants and understanding pollutant transport. In the past these models have often been used to describe particular processes and therefore remained tools predominantly used by the research fraternity. For example LEACHM (Hutson and Wagnet, 1992; Hutson and Wagenet, 1995) was developed to describe water and solute movement in the root zone. As technology has improved, pieces of the original models have been linked together into whole-of-system supermodels that enable managers to compare complex management scenarios. Examples of these supermodels used in catchment management include SWAT (Neitsch et al., 2001), HSPF (Donigian et al., 1984) and SHETRAN (Ewen et al., 2000; Nasr, Bruen et al., 2003). These models have made an extremely valuable contribution to catchment management and underpin regulatory regimes such as those based on setting Total Maximum Daily Loads for catchments in the United States.

To be effective, supermodels such as SWAT need to: (1) be computationally efficient, (2) allow considerable spatial detail, (3) require only readily available inputs, (4) be time-continuous, (5) be capable of simulating land management scenarios and (6) provide reasonable results (Arnold et al., 1998). Scaling and its impacts (i.e. dealing with heterogeneities) are directly related to these objectives as cells (rasters) or sub-catchments in the model must be defined so that the hydrological and, presumably chemical processes, can be treated as homogeneous. This is especially important as models such as SWAT are not always calibrated (Arnold et al., 1998). It follows that the sub-models that comprise SWAT are approximations that often do not have the complexity of models such as LEACHM and the output is best considered as a qualitative rather than quantitative.

In the wastewater field numerical models are also starting to make an impact. MEDLI is a computer model developed by the CRC for Waste Management and Pollution Control and the Queensland Departments of Natural Resources and Primary Industries (Queensland Department of Primary Industries and Fisheries, 2003). MEDLI models effluent production, its treatment and ultimate disposal on land and predicts the fate of water, nitrogen, phosphorus and soluble salts. Consequently, MEDLI is used for analysing and designing effluent disposal systems for intensive rural industries, agri-industrial processors (ie. abattoirs, dairy processors, user defined waste stream) and sewage treatment plants using land irrigation and would appear to be extremely valuable in that regard.

For regulatory purposes (i.e. setting phosphorus loads for wastewater irrigation sites) numerical models such as MEDLI are appealing. The input data is often readily available and the output is presented in numerical form. However, the assumptions that underpin the calculations in such models are not always clear and the errors associated with quantitative output are rarely quantified and can be significant. This is especially true of complex physico-chemical processes such as phosphorus exports from wastewater sites.

In order to retain its simplicity and flexibility, MEDLI uses a well-respected approach from HSBF requiring the user to obtain a phosphorus adsorption isotherm and initial soil phosphorus level. Phosphorus is then partitioned between the solute phase and the adsorbed phase as determined by the adsorption isotherm. The HSBF approach uses the Freundlich form of the isotherm to simulate phosphorus adsorption (Donigian et al., 1984; Johnson et al., 1984) and has not been calibrated for phosphorus storage (Ted Gardner, pers. comm. November 19 2004). These simplifications make no allowance for long-term immobilisation reactions. It follows that the predictions of the phosphorus front are considered by the developers to be conservative (i.e. the phosphorus front moves slower than predicted) and presumably phosphorus in leachate is over-predicted as a result. The more complex LEACHM model for example, partially overcomes this deficiency by using two Freundlich equations, one for the rapid adsorption processes and another for the longer-term phosphorus immobilisation. Such complexity is probably not warranted if MEDLI were used for "…designing and analysing effluent disposal systems…" (Queensland Department of Primary Industries and Fisheries, 2003) rather than for quantitatively predicting phosphorus behaviour as part of a regulatory process.

A recent study has highlighted the difficulty of using numerical models based on data from rainfall simulators for regulatory purposes. In 1999 the Alberta Intensive Livestock Stakeholder Advisory Committee requested that soil phosphorus limits be developed for Alberta, Canada (Wright et al., 2002). A Working Group integrated data from laboratory rainfall simulations, field rainfall simulations, and agricultural field scale catchments into an Edge of Field Phosphorus Export Model (EFPEM) that predicted dissolved reactive phosphorus exports based on soil test phosphorus. The working group developed interim regional load-based phosphorus limits for watersheds (i.e. drainage basins of fourth-order or larger streams) and converted these to site-specific phosphorus limits through hydrological modelling. It then translated the phosphorus concentration limits into soil test phosphorus limits through the EFPEM model.

Strong linear relationships (R^2>0.93) were found between soil test phosphorus (modified Kelowna) and dissolved reactive phosphorus (<0.45 μm) in laboratory based rainfall simulations. The equation based on the simulation data was similar to the relationship for field rainfall simulation under various crop management scenarios. Unfortunately, these relationships seriously underestimated the actual phosphorus concentrations to the extent that a Catchment Scaling Factor (CSF) of 5.9 was needed to bring the output from EFPEM within the "expected range".

Calibration of numerical models to make output in the "expected range" is a common practice and reflects the inability of many models to quantitatively represent the field situation. As demonstrated by the EFPEM model, quantitatively predicting field-scale behaviour using simulation data is extremely difficult (Nash et al., 2002). Rainfall generated by simulators has different properties to natural rainfall and due to the small area they cover, rainfall simulators rarely allow the full expression of time-dependant dissolution processes. It is not therefore surprising that the EFPEM model under-predicted dissolved phosphorus concentrations.

As our knowledge of biological systems improves, numerical models may become increasingly important for the regulation of activities such as wastewater irrigation. Such models are already useful for making qualitative comparisons of management scenarios. However, the phosphorus algorithms for most of the major models lack the specificity and process detail necessary to quantitatively predict phosphorus behaviour over the relatively small areas used for wastewater irrigation. This is not always evident as the user interface of numerical models is often extremely inviting, the assumptions underlying the calculations are not always explicit, and confidence limits are not provided for the output.

Bayesian Networks

Bayesian Networks are a tool that can be used to investigate multi-factor problems, such as phosphorus application to land in wastewater, where we have incomplete knowledge. Bayesian Networks represent uncertainty in our knowledge and use probability theory to manage uncertainty by explicitly representing the conditional dependencies between the different knowledge components. This provides an intuitive graphical visualisation of the knowledge including the interactions among the various sources of uncertainty. As a result the assumptions underlying the analyses are explicit.

Bayesian Networks are made up of a series of graphs in which the nodes represent variables, the arcs signify the existence of direct causal influences between the linked variables, and the strengths of these influences are expressed by forward conditional probabilities (Pearl, 1988).

Bayesian Networks have only recently been applied to water quality issues (Varis, 1997; Varis and Kuikka, 1999; Ames and Neilson, 2001; Ames, 2002a).For example, the new Neu-NERN Bayesian Network was developed to model the growth of algae and human effects on water quality in the Neuse estuary in North Carolina. The model was based on knowledge of the causal relationships between Dissolved Oxygen, algal blooms, hypoxia and other key variables. The links were then populated with conditional probability distributions derived from observational data, expert judgement and model simulations. The resulting network supports integrated catchment management in the Neuse Estuary by allowing decision makers to examine policy alternatives in light of risk and uncertainty. Bayesian Networks have also been used to model lake eutrophication including the spatial connectivity as well as the causal dependency between variables and control points in a water shed (Haas, 1998), and elsewhere to model the effects of fire on in-stream habitat and fish populations (Lee and Bradshaw, 1998; Ames, 2002b).

As an analytical approach Bayesian Networks have the simplicity and flexibility of indices and can provide quantitative output similar to a numerical model. However, unlike most numerical models, Bayesian Networks are easily updated as more data or new learnings become available. In addition, Bayesian Networks allow integration of qualitative information (and the associated probability) with the types of quantitative information generally included in integrated models such as MEDLI.

Some of the key benefits of a Bayesian Network approach over other existing approaches for representing natural systems are listed below (adapted from Ames, 2002a):

- In being an explicit representation of all knowledge of the system that is relevant to the decision, a Bayesian Network does not hide the assumptions and preferences that were part of its development.
- The mathematical structure underlying the Bayesian Networks graph lends itself to sensitivity analysis that can be used to determine which variables in the system have the greatest impact on outcomes of interest. By doing this, one can focus data collection on variables where both model sensitivity and model uncertainty are high.
- The modular structure of a Bayesian Network can be collapsed or expanded to add or remove extraneous variables or model elements.
- The Bayesian Network approach is data and information-centric. This allows a variety of hypotheses to be evaluated in addition to the central decision at hand.

Bayesian Networks are an extremely valuable tool for developing conceptual models of complex biological systems, such as would be required for risk based assessment of abattoir waste application sites. Once developed, the Bayesian Network can be coded for use as a component in a numerical model. As such, Bayesian Networks provide an extremely flexible modelling framework for the development of larger numerical models.

Concluding discussion

It is generally agreed that the reuse of wastewaters on land for crop and fodder production is a preferable alternative to discharging to water. Abattoir wastewaters are generally rich in nutrients such as nitrogen and phosphorus that can be substituted for fertiliser. However, the concentrations of these nutrients in abattoir wastewaters are not ideal and generally more phosphorus is applied during wastewater irrigation than is removed in produce. This is not necessarily a problem as most soils have some capacity to fix phosphorus in forms that are less susceptible to loss.

For phosphorus exports from a wastewater application site to be a problem there needs to be a source of phosphorus that is mobilised into water and transported off-site. Agronomic soil tests are one way of estimating the size of phosphorus sources. However, these tests were developed to estimate the phosphorus available to plants over a growing season rather than the phosphorus that could be mobilised in a large storm. A series of alternative tests have been developed that better predict the source factor, including the degree to which adsorption sites are occupied.

Phosphorus mobilisation often receives little detailed attention at wastewater irrigation sites. Phosphorus can be mobilised by dissolution processes that are largely controlled by soil chemical reactions and contact time, or detachment processes that are driven by kinetic energy. It is noticeable that while many of the tests to determine phosphorus accumulation are targeting dissolution processes, many of the guidelines for selecting appropriate sites for irrigation are biased towards detachment processes. For example, most guidelines regard low slopes as a good attribute for a wastewater irrigation site. Low slopes decrease the potential for detachment of sediment laden with phosphorus and the direct discharge of wastewater. However, low slopes also increase the time surface soil is in contact with water and therefore would be expected to increase the export of dissolution products from most sites.

Phosphorus transport processes are difficult to control and very difficult to quantify mathematically. Phosphorus can be transported as overland flow at the soil surface or as macropore or matrix flow through the soil profile. These processes are rarely as clears cut as such descriptions suggest.

Incorporating the source, mobilisation and transport factors into a framework that can be used to regulate phosphorus applications at wastewater irrigation sites is difficult. Wastewater applications are often based on nitrogen and soil tests are used to monitor the “environmental risk” associated with phosphorus. In the majority of cases the tests that are used, for example Olsen P and Colwell P, were not designed for this purpose. Some tests such as 0.01M $CaCl_2$ are better correlated to phosphorus dissolution than agronomic soil tests. However, sample pre-treatment (i.e. drying) would be expected to affect phosphorus pools, especially organic phosphorus.

Index systems provide a useful way of combining source, mobilisation and transport factors for management purposes. Index systems are widely used in the United States and can be incorporated into models that map the distribution of phosphorus risk. They were designed to provide broad-scale information relating to the management of phosphorus in complex biological systems and appear to have been successfully applied within the United States regulatory framework.

Numerical models are becoming increasingly popular for regulatory purposes. The user interface of such models is usually both inviting and easy to access but tends to obscure the assumptions underlying the model and the simplistic nature of the mathematical algorithms that are sometimes used. These models are particularly useful for designing and managing of wastewater irrigation systems where there are competing economic, environmental and social requirements. From a regulatory standpoint such models are useful in that they predict phosphorus exports, although the accuracy of that prediction will depend on the assumptions that are made. If such models are to be used for the regulation of abattoir wastewater irrigation sites in Australia, there is a pressing need to improve the phosphorus algorithms. Bayesian

Networks are an alternative modelling framework that may be useful as preliminary step in developing those algorithms.

Bayesian Network software offers a transparent user interface that shows the relationships between nodes (i.e. components of the model) and can allow for the complex interactions between source, mobilisation and transport factors. The design of Bayesian Networks facilitates sensitivity analyses that can be used to fine-tune a particular system and networks can be easily updated as more information becomes available. Data collected using the South Australian system “Manual for Spreading Nutrient Rich Wastes on Agricultural Land” (Clarke, 2003) system, supported by a few additional measurements, could be used for this purpose. When complete, the networks could be coded for incorporation into models such as MEDLI.

It would appear that across Australia there is a range of different standards (i.e. regulations) that apply to abattoir wastewater irrigation sites. Almost without exception, respondents from the regulatory agencies contacted through this review expressed the view that better tools were needed to assist them, especially in determining the appropriate phosphorus loadings at different sites. From an industry standpoint there is a real need to bring together and analyse the monitoring data that has been collected from abattoir wastewater sites over the years. Through mining these data the primary factors affecting phosphorus exports from these sites may be able to be identified, perhaps through a Bayesian Network, without a huge need for further long-term experimentation.

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