Section

5

5. Nutrient Transfer to the Aquatic Environment

5.1 Water movement through agricultural landscapes

Key Learning Objectives

After studying this section you should be able to:

1. Understand the key aspects of the soil water balance and appreciate how water moves from the soil profile.
2. Understand how nutrients move from agricultural systems to the aquatic environment.

Introduction

As an agricultural adviser you must be able to interpret the features of a farm's landscape and climate in order to design a fertiliser application programme that will minimize the risk of phosphorus and nitrogen movement to ground and surface waters.

Before we begin to discuss nutrient transfer to the aquatic environment, we need to understand how water moves through and off soil.

THE HYDROLOGICAL CYCLE

The amount of water on earth is more or less constant, but this water is mostly on the move. The main pathways for this movement are described as the hydrological cycle, which is shown in Figure 5.1.1. Note that the soil is involved in two major branching points in the hydrological cycle, each diverging in three ways. The first branching point is at the soil surface. There, rainfall can soak into the soil (infiltration), pond and become surface runoff, or be caught by foliage and evaporate directly (interception). Whether or not rain reaching the soil infiltrates in or runs off depends largely on how heavy the rain is, and how permeable the soil is. The permeability of a soil depends on the characteristics of the pore volume.

The second branching point in the hydrological cycle occurs within the soil. At this point, water can move laterally through the topsoil (either as 'natural' interflow through permeable soil or as induced interflow through artificial drainage systems), move on downwards out of the soil as deep drainage to become ground water, or be stored in the soil to be taken up by plants and lost as evaporation. The ability of a soil to store water depends on the pore size distribution.

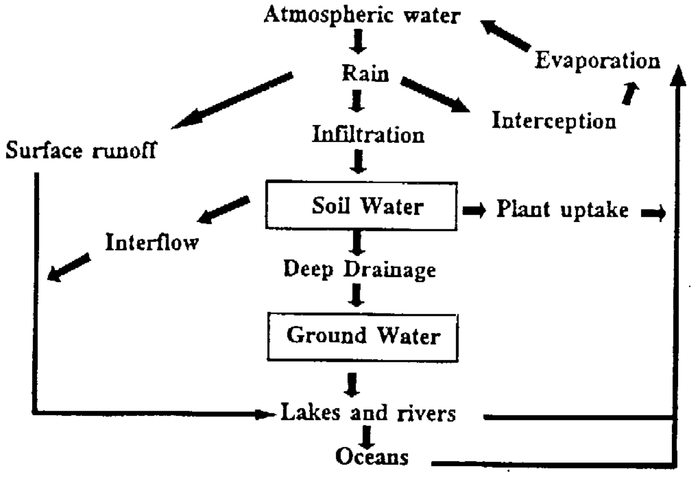

Figure 5.1.1 A simplified diagram of the hydrological cycle

A change in any part of the hydrological cycle inevitably has flow-on effects. For example, irrigation can be thought of as complementing or short-circuiting the atmospheric part of the hydrological cycle. Water from a stream or from the ground is extracted and applied to the soil. That water has to go somewhere, so inevitably there must be more plant uptake, deep drainage, and/or more interflow. Artificial drainage also changes the hydrological cycle; for example, it will enhance infiltration and reduce surface runoff.

Put more generally,

Changes in soil water storage = inputs - outputs

A change in the inputs to any part of the cycle inevitably affects the amount of storage, and/or one or more of the outputs from that part of the cycle. As an example of this simple equation, consider what typically happens to the soil under pasture near Palmerston North over a year. Rainfall provides an input of about 1000 mm. Plant uptake and evaporation are an output of about 700 mm. Over a year, the net change in storage is small, so what happens to the 300 mm excess rainfall over evaporation? It is mostly lost as deep drainage, or as artificial drainage, ending up in the ground water or the Manawatu River.

What then happens if the pasture is irrigated over summer? If the irrigation is efficient, most of the extra water will be used for evaporation and plant uptake will increase to about 840 mm. But the rest of the water applied will inevitably eventually end up either in the ground water or in streams and rivers. Of course, the nutrients (pollutants) it carries with it are as interesting as the water itself (more about this later).

We now consider *rainfall* (the main input) and *evaporation* (the main output) at the soil surface.

Rainfall

In general terms, New Zealand is fortunate in the amount of rainfall received and its reliability. But rainfall can vary considerably over quite short distances, particularly if hills or mountains are nearby. In the Manawatu catchment for example, annual rainfall can vary from 800-2000 mm/year (Figure 5.1.2). As you will see in the following discussion, climate and rainfall have a large impact on water movement and therefore nutrient loss, so it is important to have good, locally relevant rainfall data. When using rainfall information (e.g. in a water balance), data should be obtained from a site as nearby as possible. More than the average annual rainfall should be considered. In some places, rainfall tends to be winter dominant (e.g. in Whangarei, average summer rainfall is 312 mm, and winter rainfall 543 mm). Also, year to year variability should be taken into account. For example, the average summer rainfall in Feilding is 218 mm, but one year in ten it is either less than 116 mm or greater than 360 mm.

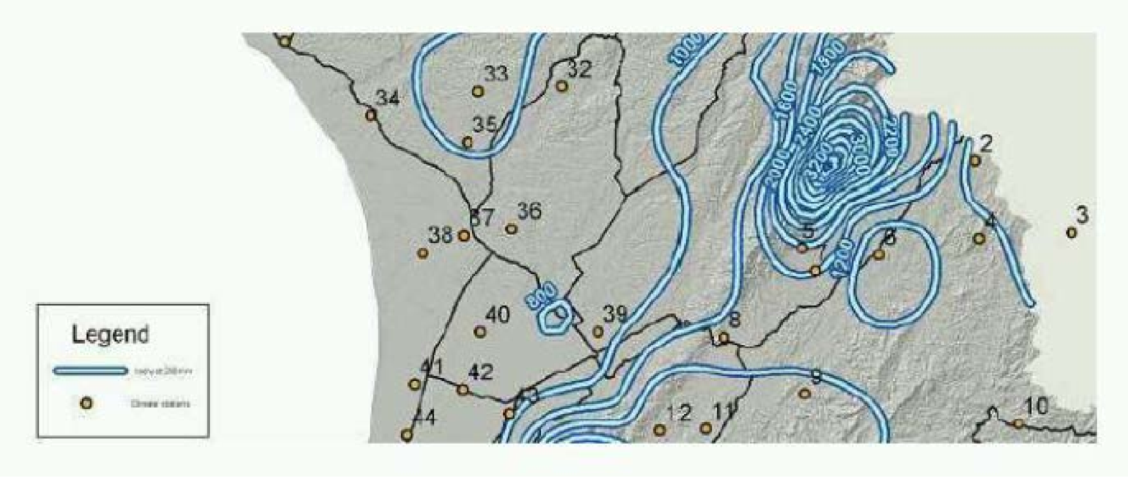

Figure 5.1.2 The isohyets of average rainfall across the Manawatu, showing the variation in rainfall in the Upper Manawatu Water Management Zone for stations 2 through 8.

Evaporation

Evaporation is the loss of water vapour from the land surface, and includes transpiration. To photosynthesise, plants absorb solar radiation. Without cooling the heat generated would destroy the plant. In excess of 95% of all water transpired by plants is used in evaporative cooling.

For evaporation to occur two inputs are needed; liquid **water**, and **energy.** The energy input provides the latent heat needed for the phase change from liquid to vapour. The evaporation rate is largely controlled by the rate of supply of whichever of these inputs is limiting. In the middle of summer, a lack of available soil water will often constrain plant growth. For a well irrigated crop, it is the energy supply that is the rate limiting factor. On a cloudy, cool day in winter, energy will be the limiting input.

As there are so many variables involved, it is often difficult to predict the evaporation. But consider,

> "evaporation from an extended area of short green crop, actively growing, completely shading the ground, of uniform height, and never short of water."

We call this the **reference crop evaporation rate**, E_r. It is called the potential evapotranspiration rate in some textbooks. Note that pasture (particularly if it is irrigated) usually fits the definition, as do a number of other agricultural and horticultural crops. Native or exotic forest will not, as they are not short. For vegetated surfaces that fit this definition, most of the energy for evaporation comes directly from sunshine landing on the leaves. Further, the reference crop evaporation rate is generally independent of the plant species involved, and the soil type. Ways of estimating the reference crop evaporation from standard weather data (e.g. sunshine hours and air temperature) exist.

The reference crop evaporation in New Zealand is a lot less variable than rainfall. For any particular month at any location, the year-to-year coefficient of variation is about 10%, compared to about 50% for rainfall (the coefficient of variation is the standard deviation divided by the mean). Thus, for irrigation and wastewater scheduling over periods of a week or more, it may be accurate enough to use long-term average reference crop evaporation values for the month in question if no other data are available. Daily reference crop evaporation rates in New Zealand vary from about 6 mm day^{-1} on a sunny summer day, to virtually zero on a cloudy winter day. Average monthly values for five locations are given in Table 5.1.1.

Table 5.1.1 Long-term average monthly reference crop evaporation (mm) (data supplied by N.Z. Meteorological Service)

Location	Auckland	Napier	P. North	Lincoln	Alexandra
January	138	151	131	137	141
February	110	117	107	105	105
March	89	92	82	75	75
April	53	57	50	42	35
May	30	33	26	18	13
June	20	22	17	11	4
July	24	24	20	13	6
August	37	38	33	25	20
September	58	63	55	50	49
October	82	99	84	86	86
November	110	126	108	111	114
December	131	145	126	133	139
Annual	882	967	839	806	787

Note that if any of the conditions specified in the definition of the reference crop evaporation are not met, the actual evaporation rate (E_a) will not equal the reference crop rate. Thus, the evaporation rate from: a shelter belt, a forest, a water-stressed crop or pasture, a bare soil with a dry surface, or a small irrigated area surrounded by dry soil will not equal the reference crop rate.

THE SOIL WATER BALANCE

Useful information about the soil water balance at a site may be obtained by plotting the mean monthly rainfall and reference crop evaporation values. This has been done in Figure 5.3 for three contrasting sites. The sites chosen are Palmerston North, Hokitika on the West Coast (for its high rainfall), and Alexandra in Otago (as it is one of the driest places in New Zealand).

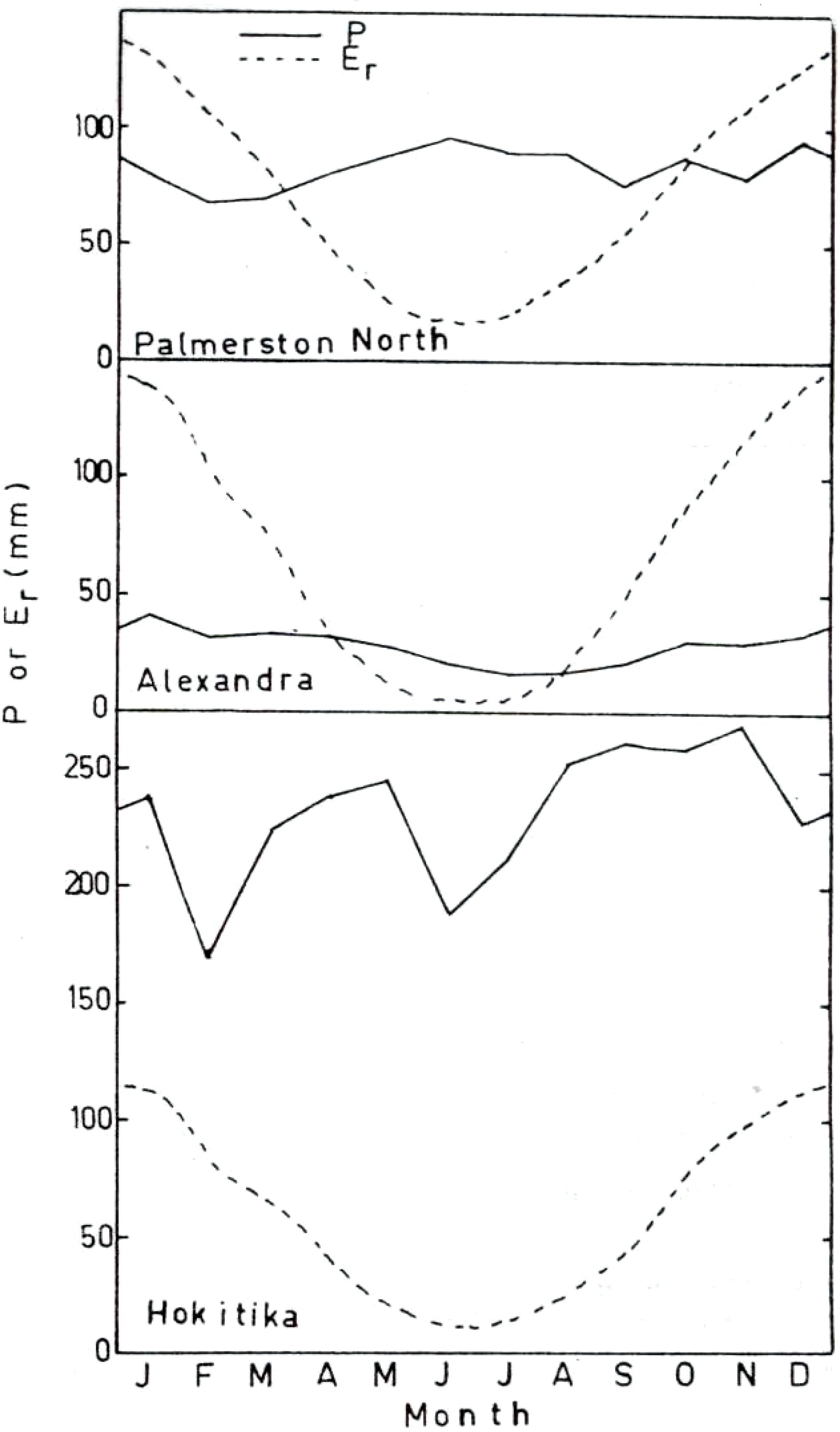

Figure 5.1.3 Average monthly rainfall (P) and reference crop evaporation (Er) at three New Zealand locations

During months when rainfall (P) is greater than the reference evaporation (E_r), either the root zone is getting recharged to "field capacity", or surplus water is leaving the root zone as deep drainage, surface runoff, interflow or in artificial drains. On the other hand, when P is less than E_r and no irrigation is applied, either the vegetation is drawing on the store of available water in the root zone, or the vegetation is water stressed and the actual evaporation rate E_a is less than E_r.

Note the very different water balances at the three locations shown in Figure 5.1.3. If we consider soils that store about 75 mm of readily plant-available water then we can speculate about the relative benefits of irrigation to farmers at the three locations.

If we apply the mass balance principle given above (that the change in storage equals the inputs minus the outputs) to the root zone over any time period, we get the soil water balance equation

$$\Delta W = P + I - R - D - S - E \qquad \text{Equation 1}$$

where

ΔW is change in water storage (mm)
P is rainfall (mm)
I is irrigated water (mm)
R is deep drainage (mm)
D is discharge from drains and/or interflow(mm)
S is surface runoff (mm)
and E is evaporation.

The soil water balance for Palmerston North is presented in Table 5.1.2.

Unless the land is steep and the surface horizon more permeable than the soil below, interflow is usually negligible. On free draining (and sometimes for artificially drained) soils, S is often negligible. Then, over winter and early spring, when rainfall periodically wets the root zone to above "field capacity", for the time period from just after one rainfall to just after the next, Equation 1 reduces to:

$$R = P - E \quad \textit{for free draining soils, or} \qquad \text{Equation 2}$$

$$D = P - E \quad \textit{for poorly drained soils with artificial drainage} \qquad \text{Equation 3}$$

If the average concentration of a nutrient or pollutant is known or can be estimated, these two equations can be used to calculate the mass of each pollutant moving in drainage or surface waters. For example, suppose the average concentration of nitrate N leaving the root zone in the drainage water is 10 g m^{-3} during a period with 85 mm of rain and 15 mm of evaporation. Then D is 70 mm or 700 m^3 ha^{-1}, so the nitrate N leaving the root zone is 7000 g or 7 kg ha^{-1}.

Table 5.1.2 Soil water balance for Palmerston North. Symbols are defined in Equation 1 except for E_r and E_a which are the reference crop and actual evaporation rates, respectively.

Month	**P mm**	**E_r mm**	**$P-E_r$ mm**	**W_A mm**	**D mm**	**E_a mm**	**I mm**
January	79	131	-52	0	-	92	39
February	67	107	-40	0	-	67	40
March	69	82	-13	0	-	69	13
April	81	50	+31	31	-	50	-
May	89	26	+63	75	19	26	-
June	97	17	+80	75	80	17	-
July	89	20	+69	75	69	20	-
August	89	33	+56	75	56	33	-
September	75	55	+20	75	20	55	-
October	88	84	+4	75	4	84	-
November	78	108	-30	45	-	108	-
December	94	126	-32	13	-	126	-
Total	995	839	+156		248	747	92

WHAT IS THE DESTINATION OF EXCESS WATER?

As discussed above, water in excess to field capacity in free draining soils or soils with artificial drainage systems exits the soil as drainage. In free draining soils, this water will end up in ground water (the soil on the left side of Figure 5.1.4). In contrast, water leaving artificial drainage systems, sooner or later, enters surface water bodies (the soil on the right side of Figure 5.1.4). There will be a number of exceptions to this rule –these relate to the generation of surface runoff. If rainfall intensity exceeds the soil's infiltration rate then runoff will result. The infiltration rate may be reduced by a number of factors. The surface soil may be less permeable because its structure has been damaged (e.g. by treading or the passage of vehicles). During prolonged or heavy rain, the capacity of the artificial drainage system may be exceeded and the water table will rise to be near the surface. As the soil surface approaches saturation, its infiltration rate will decrease. In addition, if the soil is on a slope, or the soil is water repellent then surface runoff may occur.

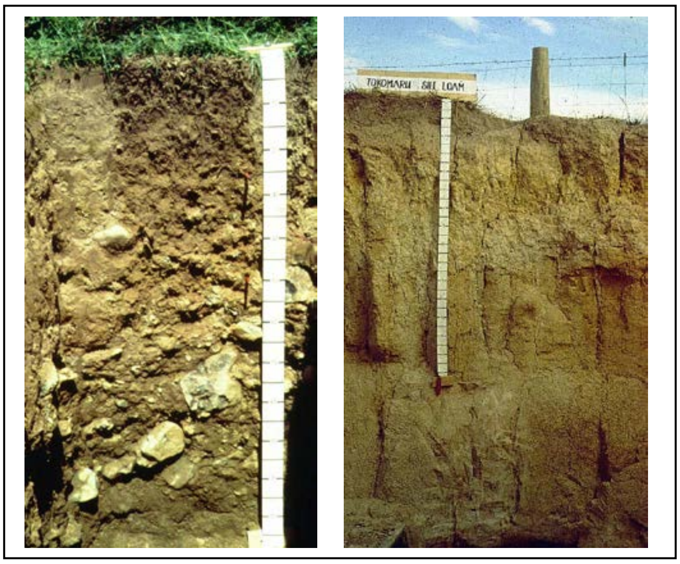

Figure 5.1.4 Examples of a free draining (left) and artificially drained (right) soils

Excess rainfall to poorly drained soils with no artificial drainage will be lost as either drainage or runoff. Where the soil is very impermeable – for example due to the presence of a fragipan – most of the winter spring rainfall will run off the profile and only very small amounts of water will drain through the profile. Whenever water moves over or through the soil, there is a risk that nutrients will move with it either in a dissolved form, or attached to soil particle, leading to the nutrient enrichment/contamination of waterways as discussed in Section 1.

OVERSEER® uses a water balance approach to provide an estimate of the total annual drainage and runoff (mm/year) generated from a farm and this figure (*under Block reports, Other values*) can be cross referenced with measured data (where it is available).

NUTRIENT MOVEMENT IN DRAINAGE AND RUNOFF

Contaminant sources are classified as: point source or diffuse (non-point) source discharges.

Point source discharges

Point source discharges are discharges of pollutants that come from a stationary or fixed exit point (e.g. from a pipe or ditch; see Figure 5.1.5). Therefore, the sources or causes of these contaminants are usually easy to locate. In terms of the discussion above about the final fate of water leaving agricultural landscapes, artificial drainage systems are a point source.

Figure 5.1.5 An example of a point source discharge from a dairy farm effluent pond.

Diffuse (or non-point) source discharges

When there is no clearly identifiable origin of the contaminants, their source is said to be 'diffuse'. Non-point source discharges are sources of contaminated water that do not have a single or specific outlet. For example, nutrients and sediment from farmland are washed into waterways by rainfall runoff or leach through the soil into ground water (Figure 5.1.6). Many of the contaminants in rivers and lakes come from urban runoff (for example stormwater and water runoff from construction sites). It is more straightforward to measure contaminant loss from point rather than diffuse sources.


Sediment rich tributaries empty
into the Waikato River

Figure 5.1.6 An example of diffuse source discharge of sediment into the Waikato River.

A comparison between the contribution of point and diffuse sources to P loss is made in Table 5.1.3.

Table 5.1.3 Estimated phosphorus fertiliser inputs and total phosphorus lost to New Zealand's freshwater from agricultural land and the human population

Nature of Discharge	**Amount of Total P (tonnes P/yr)**	
	1981[1]	**1987**[2]
Fertiliser inputs	139,000	77,000
Point Sources		
Human sewage	1,470	460
Milk processing	3,285	131
Meat processing	109	394
Cowsheds	365	657
Piggeries	1,277	525
Total	6,300	2,167
Diffuse sources		
Agricultural run-off (soil erosion, dung, fertiliser)	5,500	9,300
Agricultural industry total (Processing plus production)	10,340	11,000
Total loss to waters	11,800	11,467

Test Your Knowledge

1. Why is climate and rainfall important for water movement and why is it important that accurate rainfall, climatic and soil information is entered in to OVERSEER®?
2. What is a water balance equation and how is it used to predict drainage and runoff volumes?
3. What is the Reference Crop Evaporation rate and what assumptions is it based on?
4. How are nutrient loads calculated and what sort of approaches are used to measure nutrient concentrations in drainage and runoff?

5.2 Nutrient transport to the aquatic environment

Key Learning Objectives

After studying this section you should be able to:

1. Understand how nutrients move from agricultural systems to the aquatic environment.
2. Be familiar with the impacts of nutrient enrichment on surface and ground waters.
3. Be familiar with the sources of nutrients that are at risk to movement by water.

Introduction

Nutrients may exit the soil in a range of forms, and as was discussed in section 5.1, via two main pathways (i.e. drainage and surface runoff). Only a fraction (1 to 2%) of the total (nitrogen) N pool is present as mineral N i.e. NH_4^+ (ammonium) and NO_3^- (nitrate) at any one time. Ammonium ions tend to be somewhat immune to leaching and runoff because they are retained by cation exchange reactions or fixation by some clay minerals. In contrast, nitrate is very mobile and, consequently, prone to leaching. Studies of the impact of N movement on water quality tend to concentrate on measurements of nitrate, total N, and occasionally ammonium. As stated previously, the maximum acceptable limit for potable water is 11.3 mg nitrate -N L^{-1} (11.3 ppm).

For N leaching to occur, there needs to be an accumulation of nitrate in excess to plant requirements, and drainage. When conditions are conducive for plant growth, much of the available N is taken up and the risk of leaching is relatively low. There are three important situations where, in wet conditions, the leaching of mineral N is likely. Firstly, there are grazed pastoral systems, particularly those with high N inputs through either nitrogenous fertilisers or imported animal feedstuffs. These high N inputs are often reflected in increased stocking rates. Grazing animals concentrate and return, in dung and urine deposits, approximately 75 -90% of the herbage N that they ingest. Much of this N, particularly that in urine, is quickly converted to nitrate and is vulnerable to leaching. There may be as much as 300 – 600 kg mineral N ha^{-1} under urine patches. This difficulty is exacerbated by the preference of animals to camp and, therefore, deposit greater amounts of excreta in specific areas e.g. near watering troughs, flat ground and shelter.

The second context in which the leaching of N is probable is when the mineralization rate of N exceeds the demand of plants in late summer or autumn. As late autumn or early winter rains re-wet the soil, much of the N is likely to be flushed through the soil and move with the season's first drainage events.

Thirdly, under intensive cropping where very large quantities of nitrogen fertiliser are added, leaching can occur when the soil is bare or the crop is not yet well established.

Applications of large quantities of nitrogen fertiliser will obviously exacerbate the risk of leaching in these situations.

There have been relatively few studies of leaching losses of N under agricultural landuse in New Zealand, and what research there has been lacks consistency. For example, under sheep grazing in the Manawatu region, Field et al. (1985) measured substantial leaching of N while Heng et al. (1991) recorded relatively low losses.

Leaching losses vary both temporally and spatially. This is illustrated in Table 5.2.1 where data for nitrate movement in mole and tile drainage waters under sheep grazing are presented. Attention is drawn to the variation in winter drainage across the four years and the discrepancies in values between the two paddocks.

Table 5.2.1 Nitrate leaching loss during winter from paired paddocks (0.125 ha each) of white clover-ryegrass pasture on a mole and tile drained Tokomaru silt loam (Magesan, 1992)

	Paddock A		**Paddock B**	
Year	**Drainage (mm)**	**NO_3^--N Leached (kg ha^{-1})**	**Drainage (mm)**	**NO_3^--N Leached (kg ha^{-1})**
1988	304	13	257	9
1989	118	19	100	15
1990	260	50	235	44
1991	336	23	296	17

Generally speaking, nitrate leaching from extensive pastoral systems will be less than dairy or arable farms, with intensive market garden operations tending to leach a greater amount of nitrate. In the first instance, this reflects the disparate inputs of stocking rate and fertiliser N to these landuses. In very general terms, dairy farms and arable production systems may lose approximately 20 – 100 kg ha^{-1} yr^{-1} compared with estimates of losses of N under potato and onion crops in Pukekohe of 300 and 250 kg^{-1} N ha^{-1}, respectively.

In comparison to nitrate, (phosphoros) P is strongly adsorbed to soil constituents. Therefore only very small quantities of P are leached through the profile in drainage water. P movement to surface waters is mostly as P adsorbed to soil particles - called particulate P. However, runoff can also contain significant amounts of dissolved P, especially under pasture systems. Most runoff studies therefore seek to measure the relative size of the following fractions: particulate P, dissolved P and total P (Table 5.2.2). Sometimes a distinction is made between organic and inorganic P. Some of the P, whether dissolved or particulate, will be contained in organic compounds that probably originated in plant remains or excreta.

Particulate P makes up 75-90% of the P transported from cropping land, whereas P transport from forest or grassland is generally dominated by soluble forms (Sharpley et al., 1994) unless storm or heavy rain events are common or land slope or land surface management enhances soil erosion (Table 5.5.2). For example, with pastoral sites on hill or rolling catchments in New Zealand, particulate runoff contributes most to the form of P and N loss (e.g. Smith, 1987).

Most pasture catchment studies have highlighted the importance of stormflow in delivering particulate forms of nutrient to surface waters by surface runoff. For P, diffuse sources from agricultural land therefore comprise a high percentage of total losses from the agricultural industry (Table 5.2.2). Therefore aspects of pasture and grazing management that raise the nutrient content of surface soil or increase its susceptibility to erosion, play important roles in determining the extent of nutrient transfer to waters.

Table 5.2.2 Influence of catchment land use on drainage water quality.

Landuse	**Grazed Pasture**		**Exotic Forest**		**Native Forest**	
Catchment Output	**Range (kg/ha/yr)**	**Median (kg/ha/yr)**	**Range (kg/ha/yr)**	**Median (kg/ha/yr)**	**Range (kg/ha/yr)**	**Median (kg/ha/yr)**
Suspended solids	600-2000	1300 374[3]	300-2000	700		300[1]
Total P	0.3-1.7 1.3-2.2	0.4 1.7 0.75[3]	0.07-0.2 0.07-0.12	0.1 0.10	0.04-0.68 0.06-0.18	0.2[1] 0.12[2]
Dissolved P	0.04-0.3 0.3-0.5	0.09 0.4 0.15[3]	0.03-0.05	0.03 0.04	0.01-0.02	-[1] 0.02[2]
Total N	4-14 8.97-15.96	8 11.95 4.43	1.06-1.5	1.31	2-6 3.42-3.98	3[1] 3.67[2]
Dissolved N	1-5.3	2.2 0.77	0.07-1.3	0.5	-	0.8[1]

[1] Wilcock, 1986

[2] Cooper and Thomsen, 1988

[3] Smith, 1987

SOURCES OF NUTRIENT ENRICHMENT

In New Zealand, studies into factors affecting nutrient runoff and drainage losses have included the effect of fertilisers (Sharpley and Syers, 1976; McColl et al., 1977; Lambert et al., 1985, Gillingham et al., 1997; Gillingham and Thorrold, 2000, M^{c}Dowell et al., 2003), grazing management (Sharpley and Syers, 1979a; Turner et al., 1979; Lambert et al., 1985 Christensen et al,. 2012), slope (Sharpley and Syers, 1976), drainage type (Sharpley and Syers, 1979b, 1981), rainfall (Turner et al., 1979; Cooke, 1988; Cooke and Cooper, 1988) and agricultural effluents (Monaghan and Smith, 2004; Houlbrooke et al., 2004.)

Effects of fertiliser

Firstly, some aerially top-dressed fertiliser will fall directly into open waters. The percentage of applied fertiliser having this fate is directly related to the percentage of the top-dressed area having open water bodies. In their review of P inputs into New Zealand freshwaters, Rutherford et al., (1987) indicated that 0.5-2% of the applied fertiliser could be lost in this manner, which represents 0.05-0.6 kg P/ha/yr if topdressing rates range between 10-30 kg P/ha. Such inputs could be reduced if topdressers avoided riparian zones.

Raising Soil P Status

Fertiliser is applied to overcome nutrient deficiencies which limit pasture growth and therefore animal productivity. Thus the natural nutrient status of the topsoil is raised, which leads eventually, to increased stocking rate and activity on the surface soil.

A survey of soils of varying fertiliser history in the Wairarapa (Moir and Hedley, pers comm) shows that the traditional management of pastures with application of P and S fertilisers only, has radically increased the P status of topsoils (Figure 5.2.1) but has not led to marked increases in their N status.

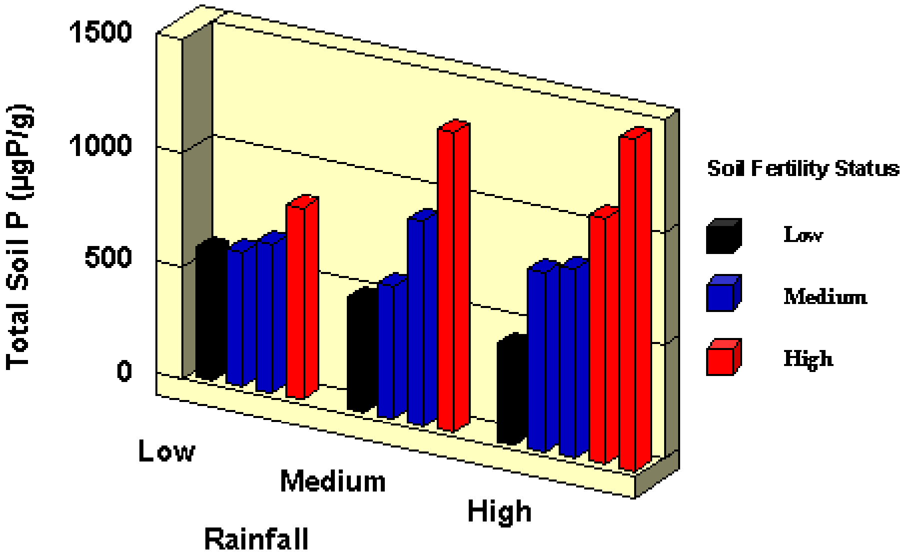

Figure 5.2.1 Total soil P concentrations in pasture soils (0-75mm) in different rainfall regimes of Wairarapa Hill Country that had received contrasting P fertiliser histories.

Thus, it is predictable that fertilised land results in relatively greater increases in the inputs of P than N into surface waters, compared with unfertilised land. This increase in total soil P increases extractable P (Olsen P soil test). Sharpley et al. (1977) demonstrated that for a series of different pasture conditions in New Zealand, there were linear relationships between extractable P in the top soil and dissolved P in the surface runoff. This is of particular importance since dissolved forms of P, rather than particulate forms are directly available to aquatic plants and therefore have greater short term impact on receiving waters. In more recent work, M^{c}Dowell and Condron (2004) showed that dissolved reactive phosphorus (DRP) in runoff and drainage generated from applying simulated rainfall to microplots can be predicted from the Olsen P test value (0-75mm) and the soil's % P retention (also known as ASC, anion sorption capacity). Soil P runoff risk predictions by Overseer® reflect this research and predicted P runoff risk varies with Olsen P value and soil type. For example, at an Olsen P of 45mg P/L, Overseer® will predict that Pallic soils with low % P retention will result in higher P losses than Allophanic soils with high % P retention. The relationships derived from microplots, however give only approximate rankings of the concentration and amounts of P carried in runoff from larger areas of hill country catchments, where storm intensity has the major impact on total P load leaving the catchment (Gillingham and Gray, 2006).

Fertiliser Form

Fertiliser form may influence the amount of P and N entering surface waters. An in-field American trial using simulated rainfall has shown that the water solubility of the P fertiliser is a good indicator of the immediate potential P loss in runoff (Shigaki et al., 2006). As the proportion of water soluble P in the fertiliser increases, it is expected that the amount of DRP in run-off from freshly fertilised land also increases. New Zealand research using turf microplots (1m long turf strips extracted from long-term fertiliser field trials) and simulated rainfall in the laboratory, has found that after P fertiliser application, the concentration of P in overland flow is higher from soils fertilised with single super phosphate (SSP) compared with reactive phosphate rock (RPR) (Figure 5.2.2), although, there is no difference 60 days after application (M^{c}Dowell et al., 2003). In field trials on dry hill country, a fresh superphosphate application had more influence on the DRP concentrations in runoff over time, than the initial Olsen P status of the soil, with highest DRP concentrations occurring within the first week of application (Gillingham et al., 1997).

Information from these New Zealand studies has been translated into recommendations for P fertiliser application, where there is runoff risk (significant slope plus high rainfall). These recommendations are included in the nutrient budgeting software Overseer®. Currently, predictions using Overseer® show a slight decline in P loss when RPR based fertilisers are used as alternatives to SSP fertiliser (Table 5.2.3), this reduction in P loss is greater when compared with SSP applied in high risk runoff months. Fertiliser form may have more impact on overall P runoff loss in the year of application and is likely to be most influential on P loss on sloping land prone to runoff.

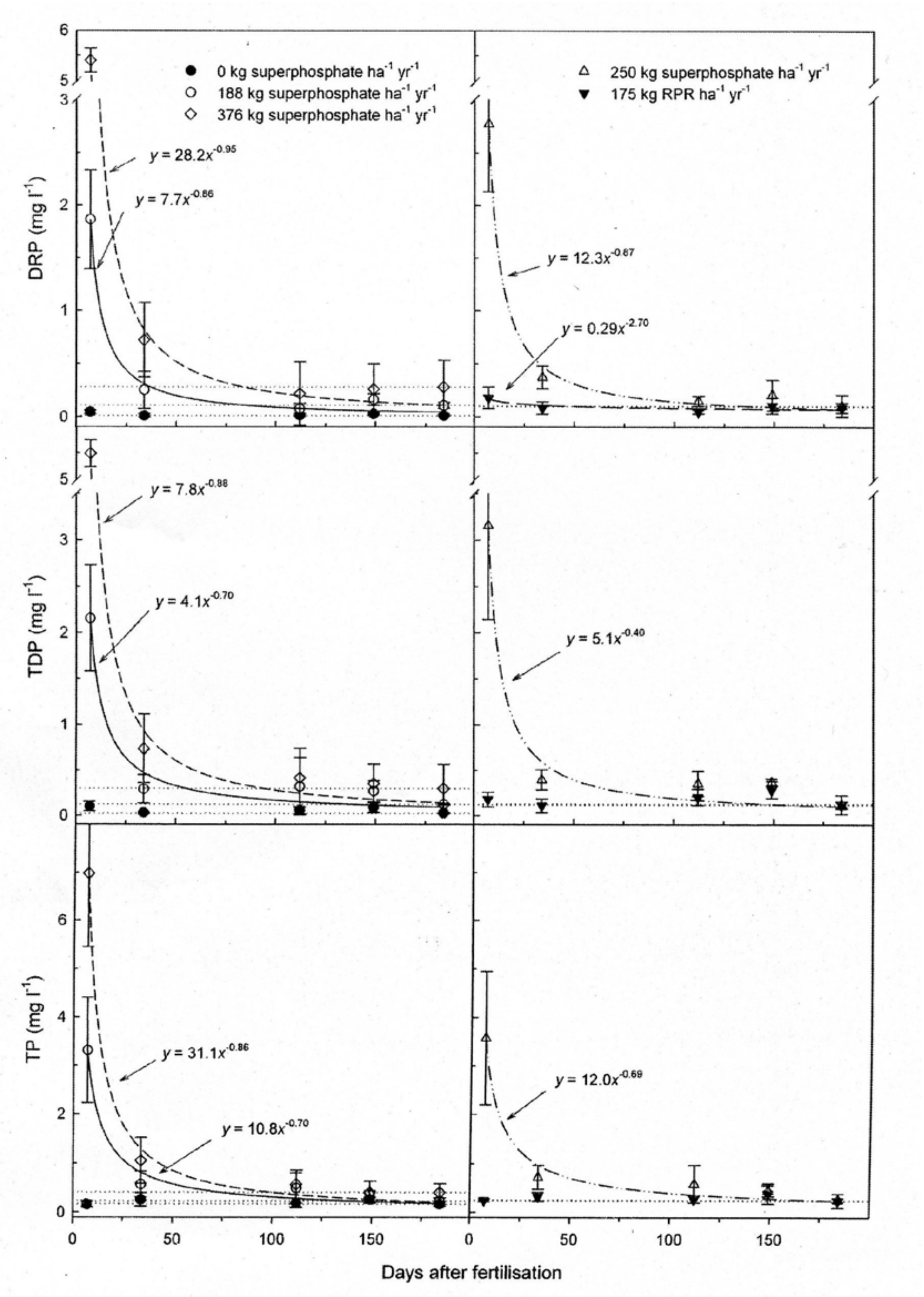

Figure 5.2.2 Concentrations of P fractions at ± 95% confidence intervals (DRP, dissolved reactive P, TDP, total dissolved P and TP, total P) with time since fertilisation. The dotted horizontal lines represent mean P concentrations before fertilisation (note 188SP (superphosphate) always lies between the 0 and 376SP treatments. All line fits are significant ($P < 0.05$) (adapted from M^cDowell et al., 2003).

Table 5.2.3 Overseer® predictions of the influence of fertiliser form and timing of application on soil and fertiliser P runoff risk and estimated P runoff loss (kg P/ha/year).

Slope	SSP						RPR		
	Applied in low run-off risk months			Applied in high run-off risk months					
	Soil	Fertiliser	P loss indices	Soil	Fertiliser	P loss indices	Soil	Fertiliser	P loss indices
Flat	Low	Low	0.1	Low	Low	0.1	Low	Low	0.0
Easy	Low	Medium	0.6	Low	High	0.8	Low	Low	0.5
Steep	Low	Medium	0.7	Low	High	0.9	Low	Low	0.5

Timing of Application

Overseer® assumes greater runoff risk during periods when the combinations of either high rainfall intensity or low evaporation tend to generate surface flow. With carefully planned timing of soluble P fertiliser application to avoid these periods, the risk of P loss can be minimised. Table 5.2.3 shows how the timing of SSP application (high compared with low runoff-risk months) can influence P loss.

If however, increasing the fertility of the soil increases plant root growth, organic matter levels and earthworm activity, soil permeability may increase leading to lower runoff volumes and increased drainage flow. The effect would be to generally reduce TP loads in runoff to streams (Lambert et al., 1985) compared with areas where runoff volumes are high. Prediction of likely amounts of P and N entering streams in catchment areas is, therefore, site specific and too complex to predict accurately and is further complicated by the management of grazing animals.

Effects of grazing management

The study of Turner et al. (1979) clearly shows the effect of grazing on accelerated artificial drainage to surface waters, particularly with respect to N. With grazing, a 14-fold increase occurred in total N losses and almost a two fold increase in P losses.

The grazing animal will have the following effects in influencing losses to surface waters:

1. The erosive force of rainfall and runoff will increase as pasture cover is removed through grazing, particularly if overgrazing occurs.
2. Pugging through grazing of the soil by animals decreases infiltration rates and increases the volume and sediment of runoff (Lambert et al., 1985; McDowell et al., 2005a).
3. Animals mobilise N and P in pasture through dung and urine return. Dung is high in soluble P and susceptible to runoff (McDowell et al., 2006).
4. The presence of animals in the stream increases erosion and deposition of excreta directly (Sharpley and Syers, 1979b). Cattle also have a propensity to enter and disturb both stream channels and saturated soil zones to a greater extent than sheep.

Lambert et al.'s (1985) study demonstrated that rotational grazing with cattle rather than sheep can cause significantly greater sediment loading and total N and P runoff. The greater impact of cattle than sheep was related to greater soil surface pugging which reduced pasture density, allowing more soil erosion. Dairying is confined to relatively flat to rolling country, yet this land use on small areas can exhibit a large effect on surface waters through point source discharge from shed wastes especially during period of low stream flow and elevated surface runoff risk from highly fertilised soils.

In summary, the published New Zealand literature on the transfer of nutrients from grazed pastures to aquatic surface waters indicates that:

1. Losses from agricultural land represent the largest land use contributor of nutrients to surface waters.
2. Relatively few studies have measured the extent of transfer losses from pastoral land to surface waters.
3. The studies conducted have generally been small scale (<20 ha) and short term (1-2 years).
4. Many studies conducted used sites with only a short history of P and N fertiliser use.
5. At farm scale level, it is impossible to identify precisely which aspects of pasture and grazing management are major contributors to nutrient runoff.

A worst case scenario for pastoral land contributing nutrients to surface waters would be: a poorly drained soil intensively grazed and of high P status on slopes that generate runoff high in sediment and adjacent to surface waters.

Effluent

Dairy and piggery effluents contain large quantities of plant nutrients. In New Zealand, these effluents can supply annually N, P and K equivalents of 17,500 t of urea, 12,500 t of single super phosphate and 28,300 t of potassium chloride, respectively, with a net fertiliser value worth 31 million dollars (Table 5.2.4). The nutrients in farm effluents can meet the N requirements of 40,500 ha of corn crop or P requirements of 62,500 ha of pasture. Application of these effluents to pasture lands has been shown to increase the dry matter yields and to enhance the nutrient status of soils (Cameron et al*.,* 1996).

Table 5.2.4 Annual fertiliser value of piggery and dairy effluents (Bolan, 1997)

Nutrient ('000 t)	**New Zealand**		**Global**	
	Piggery	**Dairy**	**Piggery**	**Dairy**
Nitrogen	2.6	5.5	213	1,125
Phosphorus	1.3	1.2	110	310
Potassium	6.8	8.6	748	2,314
Value ($ million/yr)	***7.6***	***13.5***	***964***	***3,296***

Dairy and other farm effluents are enriched with plant nutrients such as N, P and K. Biological treatment of these effluents using pond systems achieves a high degree of removal of the chemical oxygen demand (COD), the biochemical oxygen demand (BOD) and the suspended solids of the waste. The pond systems are not designed to remove nutrients that become pollutants when the effluents are discharged directly to streams.

Land treatment of farm effluents is aimed at maximising nutrient removal from effluents. However, as effluents are typically applied to only small areas of the farm (dairy farm effluent is often irrigated to approximately 15% of the farm area), the rate at which nutrients are applied is high and so there is the risk that leaching will occur following effluent application, particularly in the wet winter/spring period (Monaghan and Smith, 2004; Houlbrooke et al., 2004.). In addition, given that annual nutrient applications in effluent are often close to the permissible limit (e.g. 150 kg N/ha), nutrients will accumulate in the profile and again this will lead to an increase in the threat of enhanced loss of nutrients from effluent treatment sites.

In wet conditions, effluent irrigation may produce either surface runoff or drainage, or a combination of the two (Figure 5.2.3). This will result in effluent with very high nutrient concentrations entering waters. It is imperative that the integrity of the surface soil structure is maintained so that there is no reduction of infiltration rates and that effluent can enter the soil profile where it will hopefully be refurbished.

Figure 5.2.3 Ponding of effluent is likely to result in contamination of surface or ground waters

The merits and faults of land treatment are illustrated in Table 5.2.5 where the volume of effluent that drained through a mole and pipe system has been simulated for the past ten years. The simulation was based on the No.4 Dairy farm at Massey University, Palmerston North (soil type, cow numbers etc). A comparison is made between two scenarios; direct discharge to a stream and land application. Irrigation of effluent successfully reduces the volume of effluent entering surface waters from 6806 m^3 to 0 – 1750 m^3 per annum. It is noteworthy that much of this reduction in effluent entering the stream was affected in the summer period, when the stream would be most sensitive to the impacts of nutrient contamination, due to low flow. However, during the drainage season (variable in duration but typically winter to early summer) approximately 80% of the irrigated effluent drains to surface water.

Table 5.2.5 The results of a simulation exercise that compared the volume of effluent that enters surface water directly (direct discharge from the second pond) with that which drains following land application to a mole and pipe drained soil (Tokomaru silt loam). The proportion of effluent that drains during the drainage season is also shown.

Year (lactation season)	Drainage and runoff of rainfall (mm)	Effluent exiting outlet of pond system (m^3)	Drainage of effluent from land application - (m^3)	Proportion of effluent irrigated in drainage season that drains (%)
1991 - 1992	45	6806	1016	38
1992 – 1993	154	6806	1750	78
1993 – 1994	0	6806	0	0
1994 - 1995	210	6806	1774	64
1995 – 1996	233	6806	2351	75
1996 – 1997	106	6806	1226	58
1997 – 1998	60	6806	1456	72
1998 – 1999	364	6806	1919	78
1999 – 2000	0	6806	230	11
2000 – 2001	158	6806	1529	75

Measuring nutrient loss

WHY BE QUANTITATIVE?

When we look at a river or lake we can all make a value judgement as to whether it looks 'clean and natural' or 'ugly and polluted'. There is no need for us to quantify or put a number on our feelings in order to make such a judgement.

Similarly, as we have seen, the Resource Management Act (1991) usually describes water quality in very general narrative terms. For example, if the community wishes to use a lake or river for swimming then the Third Schedule of the Act suggests that:

> *(i) The visual clarity of the water shall not be so low as to be unsuitable for bathing.*
>
> *(ii) The water shall not be rendered unsuitable for bathing by the presence of contaminants.*
>
> *(iii) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.*

These all sound very sensible - I am sure we would all agree with them. And there is not a single number in sight!

It is only when we look into these narrative standards in more detail that some questions arise. In point (ii), what exactly do we mean by 'the presence of contaminants'? What contaminants are we talking about? And what do we mean by 'presence'? Do we mean that there should be ABSOLUTELY NO contaminants in the water? Such very pure water is hard to obtain. Or do we mean, more realistically, that some contamination might be acceptable as long as it is at a low level that has been demonstrated as 'safe'? If this latter statement is really what we mean, then we have to define what this 'safe' level is - usually with a number. We also have to monitor the water to make sure that this 'safe' number is not exceeded. This usually involves a quantitative chemical, physical or biological analysis. Finally, if we want to manage our water in a sensible way, we must be able to predict.

Imagine a big industry applies to a Regional Council to discharge waste material into a river. The Regional Council is not sure whether the discharge will increase the contaminant levels in the river above the 'safe' level we discussed earlier. What is the Regional Council to do?

One approach would be to allow the industry to discharge the waste and then to monitor the river downstream from the discharge point. If the 'safe' level of contaminants was exceeded then the industry would have to stop discharging.

A monitoring approach such as this is effective - but very inconvenient. Firstly, we had to damage the environment to find out whether the discharge was 'safe' or not. Secondly, we have probably created a lot of uncertainty (and cost) for the industry.

How much easier it would be if we could predict what the effect of the discharge on the river would have been. In fact, this can be done, but it does involve some calculations and the use of numbers. We will explore this now.

VOLUME AND WEIGHT

When managing water quality there are two important concepts to grasp.
These concepts are:

- Volume of water
- Weight of material

They are discussed in more detail below.

VOLUME

In environmental matters the amount of water in a system is usually defined by volume (rather than weight). The volumes involved vary enormously. Thus, an environmental scientist may carry out tests on a few drops of water, whereas a catchment manager may be dealing with enormous rivers such as the Clutha or the Waikato. When people are dealing with large volumes of water, they normally measure them in 'cubic metre' which is abbreviated as m^3. As the name suggests, 1 m^3 is the volume enclosed by a box 1 metre (m) long, 1 m wide and 1 m high. If you have a metre rule, mark out what this would look like in a corner of the room. It is quite large.

1 m^3 of pure water weighs almost exactly 1 tonne.

If we are dealing with smaller volumes of water we use a smaller unit- a litre (L). This is a unit that you will all be familiar with. In many parts of the country, milk can be purchased in 1 L bottles. If we were to take the 1 m^3 box mentioned earlier and fill it up with water using a 1 L container, it would take exactly 1,000 1 L containers of water to fill the box. We can write this as:

$$1\ m^3 = 1000\ L$$

or, in words:

one cubic metre has the same volume as 1000 litres.

When we are dealing with even smaller volumes, we use a unit called a millilitre (ml). Once again, this is a unit that you will be familiar with when cooking or dispensing medicine to children.

If you filled a 1 L milk container with water using an eyedropper that could take 1 ml, you would find that 1 L contains exactly 1000 ml. Or:

$$1\ L = 1000\ ml$$

Now, we know that:

$$1\ L = 1000\ ml \text{ and } 1\ m^3 = 1000\ L$$

so:

$$1\ m^3 = 1{,}000 \text{ X } 1{,}000\ ml$$

$$= 1{,}000{,}000\ ml$$

in words:

one cubic metre has the same volume as one million (1,000 x 1,000) millilitres.

If:

$$1\ m^3 = 1{,}000{,}000\ ml$$

Then:

$$2\ m^3 = 2{,}000{,}000\ ml$$

and:

$$10\ m^3 = 10{,}000{,}000\ ml$$

Conversely going back the other way:

$$3{,}000{,}000\ ml = 3\ m^3$$

and:

$$4{,}000\ ml = 4\ L$$

It is very important that you have confidence in converting volumes from millilitres to litres and cubic metres and back again.

You should study the Worked Examples below and satisfy yourself that you understand the concepts.

Index Notation

As you have already seen, environmental matters span the range of volumes from the very small (ml) to the very large (m^3). When dealing with the large range in scale, it is inconvenient to write numbers like 10 million out in full, as 10,000,000. Instead we use an index notation as follows:

$$\begin{aligned}
10 &= 10 \text{ x } 1 &&= 10^1 \\
100 &= 10 \text{ x } 10 &&= 10^2 \\
1{,}000 &= 10 \text{ x } 10 \text{ x } 10 &&= 10^3 \\
10{,}000 &= 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 &&= 10^4 \\
100{,}000 &= 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 &&= 10^5 \\
1{,}000{,}000 &= 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 \text{ x } 10 &&= 10^6
\end{aligned}$$

In a similar way:

$$1 \div 10 = \frac{1}{10} = 10^{-1}$$

$$1 \div 100 = \frac{1}{10 x 10} = 10^{-2}$$

$$1 \div 1{,}000 = \frac{1}{10 x 10 x 10} = 10^{-3}$$

Most of you will be very familiar with this notation from school or university. If you are not certain about it, we suggest you get a simple book on mathematics out from your local library and work through the appropriate exercises.

Generally, the rule is that when multiplying numbers the indices are added: e.g.

$$100 \text{ x } 100 = 10{,}000$$

is the same as:

$$10^2 \text{ x } 10^2 = 10^4$$

or

$$1{,}000 \text{ x } 1{,}000 = 1{,}000{,}000$$

is the same as:

$$10^3 \text{ x } 10^3 = 10^6$$

When dividing numbers, the indices are subtracted: e.g.

$$1000 \div 10 = 100$$

is the same as

$$10^3 \div 10^1 = 10^2$$

or

$$1{,}000{,}000 \div 1{,}000 = 1{,}000$$

is the same as

$$10^6 \div 10^3 = 10^3$$

WEIGHT

The units of weight in common usage in environmental matters are all based on the 'gram' (g). This once again, is a unit you are familiar with. One millilitre of water weighs 1 g. A tub of margarine may weigh 500 g.

As with volumes of water, the weights of material encountered in environmental matters span an enormous range. The amount of sediment carried by a river may be measured in tonnes, whereas a highly toxic chemical such as dioxin may be of concern in quantities ranging from a thousandth to a millionth of a gram.

The main units of weight you will encounter are as follows:

microgram (μg) = 1 millionth of a gram
milligram (mg) = 1 thousandth of a gram
gram (g) = 1 gram
kilogram (kg) = 1 thousand grams
tonne (t) = 1 thousand kilograms (1 million grams)

As with volumes, you should be proficient at converting weights from grams to kilograms and tonnes and back again. The worked examples and exercises will assist you with this.

FLOW

Now that you are familiar with volumes and weights we can employ these units in a range of environmental measurements. The first of these is **flow**.

The flow of water (in a river or pipe) is the volume of water passing a point in a given time - usually 1 second.

In a pipe, the flow of water is also the quantity discharged from the end of the pipe in a given time. In rivers, the flow is usually measured as the number of cubic metres per second or m^3 /sec. These are also referred to as cumecs. Thus, if you stand on the bank of a river and 50 m^3 of water goes past you every second, the river would have a flow of 50

cumecs. Flow in pipes is usually much smaller and is commonly measured in litres per sec or L/sec
Just as with volume:

$$1\ m^3 = 1000\ L$$

so with flow

$$1\ m^3/sec\ (cumec) = 1000\ L/sec$$

You must be confident at converting flows from cumecs to litres per second and back again. There are worked examples and exercises later in the Topic to help you with this.

CONCENTRATION

The **concentration** of a material in water (or air) usually refers to the **amount** of that material in a given **volume** of water (or air). The amount of material is usually (but not always) expressed by **weight.** Thus the concentration of a contaminant, such as P, in a river will be expressed as the number of grams of P per cubic metre (gm^3) of water.

Now we come to a tricky bit!

Imagine that we have a 1 m x 1 m x 1 m container that is filled with water. The volume of water will be 1 m^3. If, dissolved in this water, was 1 g of P, then the concentration of P would be 1 g/m^3. This P would be distributed uniformly throughout the water.

We know from an earlier section that:

$$1\ m^3 = 1000\ L$$

So if we were to remove 1 L of water from the container then we would be removing one thousandth of the total volume of water. We would also be removing some P in the water. As the dissolved P is distributed evenly through the water, one thousandth of the original volume of water would contain one thousandth of the original weight of P, originally 1 g.

Once again, from a previous section, we know that one thousandth of a gram is a milligram (mg). So a solution with a concentration of P of 1 g/m3 also contains 1 mg/L of P. By a similar calculation we could show that the same solution would contain 1 μg/ml.

In summary:

Concentration = weight/volume

and a concentration expressed as g/m^3
is the same when expressed as mg/L
and the same when expressed as μg/ml

NOTE: We have seen earlier that 1 ml of pure water weighs 1 g and that 1 m^3 of water contains 1,000,000 (1×10^6) ml.

Therefore 1 m^3 of water weighs 1,000,000 g

So, a concentration of 1 g/m^3 of P contains 1 g of P in 1,000,000 g of water. This concentration is sometimes termed 1 part per million (ppm).
Concentrations expressed as 'parts per million' are very common in environmental literature.

A solution that has a concentration of 1 ppm will have a concentration of:
1 g/m^3
and as we have said before 1 mg/L
and 1 μg/ml

Although a concentration of 1 ppm is very dilute, concentrations of many environmentally damaging materials in natural water systems may be very much less than this. Perhaps even less than one thousandth of one ppm. A concentration that is one thousandth of one ppm is called one part per billion (ppb).

Thus: 1,000 ppb = 1 ppm

A solution that has a concentration of 1 ppb will have (a concentration of):
1 mg/m^3
and 1μg/L

You need to be comfortable dealing with chemical calculations expressed in a wide variety of different units.

Concentrations of sediment

Until now, the concentrations we have discussed have all involved volumes that move in multiples of 1,000 i.e. millilitres, litres and cubic metres. Unfortunately, for reasons that are buried in the past, the concentration of sediment in water is usually expressed as the number of grams of sediment in 100 millilitres of water (or percent). Thus water with a sediment concentration of 1 percent has 1 g of sediment suspended in 100 ml of water.

Concentrations of bacteria

When dealing with chemicals or sediment, the weight of the material dissolved or suspended in the water is a useful measurement. When dealing with contamination of water by bacteria, it is the number (rather than the weight) of bacteria that is more important. As with sediment, concentrations of bacteria are expressed per 100 ml.
Thus the concentration of enterococci bacteria in water is expressed as:
(number of bacteria)/100 ml

You will see numbers such as this quoted in water quality standards and also in newspapers in summer when reporting on water quality at local beaches.

FLUX AND LOAD

Water is the vector that is responsible for much of the losses of nutrients, sediment, and/or bacteria from the soil. When the chemical is dissolved, it is referred to as a solute. Flux is a concept that is commonly employed in science to describe the rate at which mass or energy moves. So the emission of greenhouse gases, P runoff or the leaching of pesticides all have a flux. Very simply the flux is the quantity of material that passes through a unit area in a unit time. If we think about N leaching then the most practicable units are; kg for the quantity of N; a hectare for the area and we are often interested in annual losses. If you are told that the N loss from a farm in 30 kg N ha^{-1} yr^{-1} then this is the flux of N loss. Sometimes, the flux is referred to as the load i.e. these can be used interchangeably (when talking about nutrient flow in rivers, load is probably the preferred term – see below).

The flux (q_s)is given by the equation:

$$\mathbf{q_s = q_w \times c}$$

Here q_s = solute flux (this relationship also works for material that isn't strictly speaking dissolved)
q_w = water flux (remember that water is conveying the solute) and
c = concentration.

This is a very important formula to remember.

Given the following data, we can calculate the rate at which N is lost from this paddock i.e. the flux.

Annual drainage = 300 mm (i.e. 0.3 m yr^{-1})
Average N concentration of this drainage water = 8 g m^{-3}
What is the flux or rate that N moves through the unit area (ha) in the unit time (yr)?
Firstly we need to identify q_w – the water flux i.e. the amount of water that is draining from a ha each year (and taking the N with it).

$$q_w = 10{,}000\ (m^2\ ha^{-1}) \times 0.3\ (m\ yr^{-1}) = 3000\ m^3\ ha^{-1}\ yr^{-1}$$

(Remember that there are 10,000 m^2 in each ha and that 300 mm yr^{-1} = 0.3 m yr^{-1}.)

So the rate of N loss or flux q_s is:

$$q_s = 3000 \times 8 = 24000\ g\ ha^{-1}\ yr^{-1}$$

As stated above, for N leaching losses, kg is the preferred unit so we express this as 24 kg N ha^{-1} yr^{-1}. You should check that you understand how these units were arrived at.

For the loads of nutrient in rivers we have an identical process

Load = Flow x Concentration

Thus for **chemicals**; if the flow in a river is 50 m^3/sec (cumecs) and the concentration of a chemical such as P is 15 mg/m^3 (15 ppb) then the load of P is given by:

Load = Concentration x Flow

$$= 15\ mg/m^3 \times 50 m^3/sec$$
$$= 750\ mg/sec\ of\ P$$

REMEMBER: *Before calculating a load, you must make sure that the volume unit (usually L or m^3) used to describe the flow is the same as the volume unit used to describe the concentration.*

WHY CALCULATE NUTRIENT LOADS IN RIVERS?

The concept of a contaminant 'load' is very important because it enables us to calculate the effect of a waste discharge on a river system. Take the river described in the previous section upstream of a sewage discharge. Imagine that the discharge has a flow of 50 L/sec (0.05 m^3/sec) and a P concentration of 4 g/m^3.
What will be the concentration of P in the river downstream of the discharge?

To calculate this we must make use of loads - we CANNOT just add or average the concentrations! When we think about it - this makes sense. The effect of a discharge will depend both on how dirty it is (concentration) and how much there is of it (flow).

As we saw in the previous section, the load of P in the river upstream of the discharge was:

Upstream River Load = 750 mg/sec = 0.75 g/sec of P

The load of P in the discharge (i.e. the amount coming out of the pipe) is:

Discharge Load = Concentration x Flow

= 4 g/m^3 x 0.05 m^3/sec
= 0.2 g/sec of P

The load downstream of the discharge will be the sum of the load in the river initially (i.e. upstream of the discharge) and the load of P
coming out of the pipe i.e.:

Downstream River Load = Upstream Load + Discharge Load

= 0.75 g/sec + 0.2 g/sec
= 0.95 g/sec of P

In the same way:

Downstream Flow = Upstream Flow + Discharge Flow

$$= 50 \text{ m}^3/\text{sec} + 0.05 \text{ m}^3/\text{sec}$$
$$= 50.05 \text{ m}^3/\text{sec of water}$$

Now that we know both the concentration and the flow downstream of the discharge, we can use the equation:

Load = Concentration x Flow

in a different way. Rearranging the equation gives us:

Concentration = Load/ Flow

So, downstream of the discharge:

Concentration = Load/ Flow

$$= (0.95 \text{ g/sec})/(50.05 \text{ m}^3/\text{sec})$$
$$= 0.019 \text{ g/m}^3 \text{ of P}$$
$$= 19 \text{ mg/m}^3 \text{ (or 19 ppb) of P}$$

Therefore, the discharge has increased the **Concentration** of P in the river from 15 mg/m^3 to 19 mg/m^3.

WORKED EXAMPLES

1. Express the following volumes as cubic metres (m^3)

(a) 750 L

(b) 650 ml

(a)1 m^3 contains 1,000 L

$$\text{Therefore: } 750 \text{ L} = \frac{750}{1000}$$
$$= 0.75 \text{ m}^3$$

(b)1 m^3 contains 1,000,000 ml

$$\text{Therefore: } 650 \text{ ml} = \frac{650}{1000000}$$
$$= 0.00065 \text{ m}^3$$
$$= 6.5 \times 10^{-4} \text{ m}^3$$

2. Express the following volumes as litres (L)

(a) 1.65 m^3

(b)3,400 ml

(a)1 m^3 contains 1000 L

$$\text{Therefore: } 1.65 \text{ m}^3 = 1.65 \times 1000$$
$$= 1{,}650 \text{ L}$$

(b)1 L contains 1,000 ml

Therefore: 3,400 ml = $\frac{3400}{1000}$
= 3.41

3. Express the following weights in kilograms (kg)

(a) 850 g
(b) 1.2 tonnes (t)

(a)1 kg contains 1,000 g

Therefore: 850 g = $\frac{850}{1000}$
= 0.85 kg

(b)1 t contains 1,000 kg

Therefore: 1.2 t = 1.2 x 1,000
= 1,200 kg

4. Express the following flows in m^3/sec (cumecs)

(a) 75 L/sec
(b) 2,300 L/sec

(a)A flow of 1 m^3 /sec is equivalent to 1000 L/sec

Therefore: 75 L/sec = $\frac{75}{1000}$
= 0.075 m^3/sec (cumecs)

(b)A flow of 1 m^3 /sec is equivalent to 1000 L/sec

Therefore: 2,300 L/sec = $\frac{2300}{1000}$
= 2.3 m^3/sec (cumecs)

5. If a solution has a P concentration of 13 parts per billion (13 ppb), what is the P concentration expressed in parts per million (ppm)?

From the study material, we know that: 1 ppb = 1 mg/m^3 and 1 ppm = 1 g/m^3

So a concentration of 13 ppb is 13 mg/m^3. There are 1000 mg in a gram.

Therefore: 13 ppb = $\frac{13}{1000}$
= 0.013 g/m^3
= 0.013 ppm

6. **Effluent from the Palmerston North sewage works is discharged into the Manawatu River. If the discharge from the sewage works has a flow of 60.0 litres/sec and a concentration of dissolved P (DIP) of 4.00 g/m^3 and the river upstream of the discharge has a flow of 95 m^3/sec and a DP concentration of 0.009 g/m^3 then calculate:**

   (a) The load of DP in the sewage effluent.
   (b) The load of DP in the river upstream of the discharge.
   (c) The load of DP in the river downstream of the discharge.
   (d) The concentration of DP in the river downstream of the discharge.

Flow from sewage works = 60.0 L/sec
= 0.06 m^3/sec

(a) Load of DP in sewage effluent
= flow x concentration
= 0.06 x 4
= 0.24 g/sec

(b) Load of DP in river upstream of discharge
= 95 x 0.009
= 0.855 g/sec

(c) The load of DP in the river downstream of the discharge
= 0.24 + 0.855
= 1.095 g/sec

(d) Flow of river downstream of discharge
= 95 + 0.06
= 95.06 m^3/sec

Concentration of DP in the river downstream of discharge
= 1.09 ÷ 95.06
= 0.0115 g/m^3

7. **A dairy farm loses 250mm (0.25m) in drainage below the root zone in a typical year. If the concentration of N in drainage water is 10 ppm calculate:**

   (a) The total amount of water draining from 1 ha of this farm.
   (b) The annual per hectare loss (load) of N (kg N/ha) from this farm in drainage water.

Quantity of water draining from 1ha:
1 ha = 10,000m^2
10,000m^2 x drainage depth (0.25m)
= 2,500 m^3
= 2,500,000L (1 m^3 = 1000 L)

Annual per hectare loss (load) of N in drainage water
Load = Volume (Flow) x Concentration
1 ppm N = 1 g N/m^3
10 ppm N = 10 g N/m^3
N Load = 2,500 m^3/ha x 10 g N/m^3
= 25,000 g N/ha
= 25 kg N/ha

Measuring nutrient loss from agricultural land

Obtaining the information required to accurately and reliability determine nutrient load, is not always easy. Measuring the volume of water leaving a catchment or sub-catchment requires expensive equipment and the concentration of nutrients in the water need to be measured at a frequency which captures the changes in concentration. For example, some nutrients such as P tend to accumulate over the drier periods and a flush of P can occur in the first runoff event of the season, resulting in elevated P concentrations in the water samples. Thus, it is important to accurately measure the changes in nutrient concentration across the season in order to generate reliable data. Similarly, the quantity of drainage water is likely to vary considerably over the year, so it is important that each drainage event is measured accurately.

MEASURING NUTRIENT LOSS IN DRAINAGE

Where artificial drainage is installed, the volume leaving the outlet can be measured year round using a flow meter. Drainage can also be captured at a smaller scale using lysimeters. A lysimeter is a physical barrier (usually a cylinder) that is placed in the soil, in order to measure the drainage water that moves through the soil profile. Lysimeters can range in diameter from 10 cm to several metres (Figure 5.2.4), with their depth often varying in relation to the diameter and the characteristics of the soil profile. Lysimeters can be collected in the field and then transported back to the laboratory or an outside lysimeter setup, or they can remain *in situ*.

Figure 5.2.4 Examples of lysimeters, how they are collected and their range in size.

A commonly known lysimeter study is that at Lincoln University, where lysimeters are placed in paddocks intensively grazed by dairy cows. Drainage water is collected below the lysimeters (Figure 5.2.5) and the volume of drainage and the concentration of nutrients in the drainage water is measured. Lysimeters are commonly used to assess the impacts of different treatments (i.e. fertiliser, urine, DCD applications or a combination these) on nutrient loss.

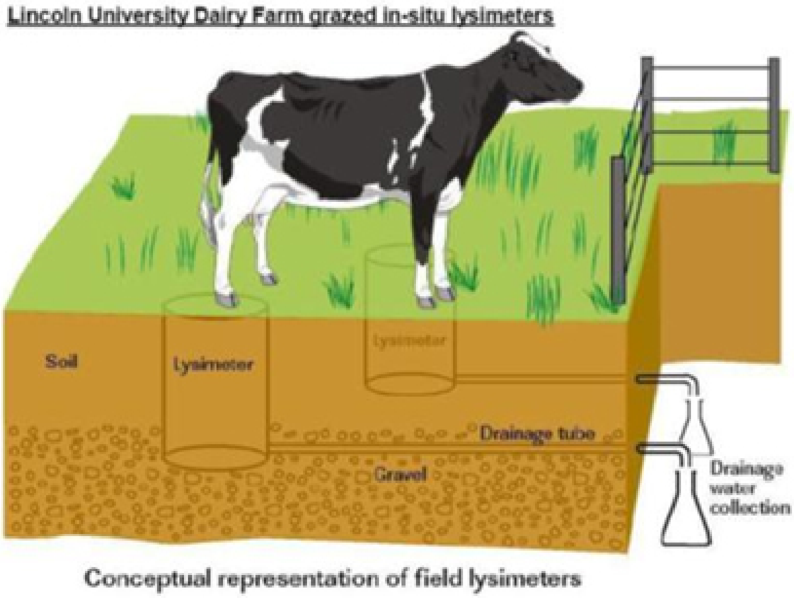

Figure 5.2.5 Examples of in-situ filed lysimeters at Lincoln University and how the drainage water is collected in order to measure drainage volume and nutrient concentration

Where artificial drainage is installed in the form of mole or tile drains, drainage water can be collected and volume measured using a tipping bucket method (Figure 5.2.6). The tipping buckets are calibrated and the volume of water can be measured by the number of times the bucket fills and tips over. The concentration of nutrients in the drainage water is measured throughout the drainage event to allow nutrient load to be calculated.

Figure 5.2.6 Drainage water being collected from artificial tile drains installed in a pasture paddock at No. 4 dairy, Massey University, using a tipping bucket method. (Photo: C Christensen).

Measuring nutrient loss in surface runoff

Surface runoff is more difficult to measure as it is often generated from large areas of land in relatively short time frames (i.e. during storm events), so it is essential to use in-field instrumentation in order to measure flows and collect representative water samples which can later be analysed for nutrient concentration. Surface runoff can be measured at scales ranging from the whole of catchment scale (in this case, water flows and water samples would be measured at various points along the river draining the entire catchment), to sub-catchment scale (a smaller catchment area which drains in to a small creek or tributary would be measured), to the plot scale (an area may be hydrologically isolated using physical barriers such as wood or existing land features in order to collect runoff from a smaller land area) (Figure 5.2.7).

Figure 5.2.7 Examples of sub-catchments (left) and artificial sub-catchment (right) used to measure nutrient loss in surface runoff.

In order to generate more controlled and reliable surface runoff, rainfall simulation can be used. Due to the costs and logistics involved, rainfall simulation is generally undertaken at much smaller scales, however it can be a useful technique to measure the effect of treatments (i.e. fertiliser or effluent application rate and timing on different soil types) on nutrients lost in surface runoff. Figure 5.2.8 shows a large scale rainfall simulation experiment established in the Waikato hill country to measure bacteria loss in surface water runoff.

Figure 5.2.8 Large scale rainfall simulation experiment established in the Waikato hill country to measure bacteria loss in surface water runoff (Photo: R Collins).

Test Your Knowledge

1. Describe what form and by what process, nitrogen is most commonly lost to surrounding water ways?
2. What farm management factors increased the risk of nitrate loss to the environment?
3. What is the most common process of P loss in New Zealand?
4. What farm management factors increase the risk of P loss to the environment?
5. If you wanted to measure the effect of different N concentrations in urine on the amount of nitrate lost in drainage water, what approach might you use?

Revision of calculations

1. Complete the horizontal rows (b-d) in the following table. Each entry in a row represents the same volume, but expressed in different units. Row (a) provides an example

	Millilitre (ml)	**Litre (L)**	**Cubic metre (m^3)**
a	1,300	1.3	0.0013
b			0.85
c		27	
d	6,300		

2. Complete the horizontal rows (b-d) in the following table. Each entry in a row represents the same weight, but expressed in different units. Row (a) provides an example.

	Microgram (μg)	**Milligram (mg)**	**Gram (g)**	**Kilogram (kg)**	**Tonne (t)**
a	4.0×10^6	4.0×10^3	4.0	4.0×10^{-3}	4.0×10^{-6}
b	7,300				
c				0.72	
d			3,700		

3. Complete the horizontal rows (b-d) in the following table. Each entry in a row represents the same concentration but expressed in different units. Row (a) provides an example.

	Percent (% or g/100 ml)	**Parts per million (ppm or μg/ml or mg/L or g/m^3)**	**Parts per billion (ppb or μg/L or mg/m^3)**
a	0.045	450	4.5×10^5
b			820
c		15	
d	0.65		

4. Imagine a situation described in Worked Examples 10 and 11, whereby a pipe is discharging effluent into a river. The effluent contains dissolved phosphorus (DIP).

The table below provides data for three Scenarios (a-c).

	Scenario (a)	**Scenario (b)**	**Scenario (c)**
River flow upstream of Discharge	45 m^3/sec	87 cumecs	55 m^3/sec
Discharge flow	40l/sec	0.067 m^3/sec	0.082 cumecs
DP concentration in river Upstream of discharge	0.012 g/m3	0.016 mg/l	14 ppb
DP concentration in discharge	4.2 ppm	3.5 μg/ml	4.8 ppm

Answer the questions below for each of the scenario's (a), (b) and (c):

The load of dissolved phosphorus (DP) in the effluent discharge.

(ii) The load of (DIP) in the river upstream of the discharge.

(iii) The load of DP in the river downstream of the discharge.

(iv) The concentration of DP in the river downstream of the discharge.

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5.3 Best Management Practices for Protecting the Aquatic Environment

Key Learning Objective

After studying this section you should be familiar with the farm management practices that help minimise nutrient movement to the aquatic environment.

Introduction

In the previous section you studied some of the pathways by which nutrients move into the aquatic environment, including an appreciation of some of the nutrient sources that enrich soil water. We will now consider some of the options available for managing the problem of water contamination by nutrients from agriculture.

TARGET APPLICATION OF FERTILISER

Differential aerial application

Current research is developing new digital mapping technology that will improve the definition of spatial variability in fertiliser requirements of different land units (Figure 5.3.1).

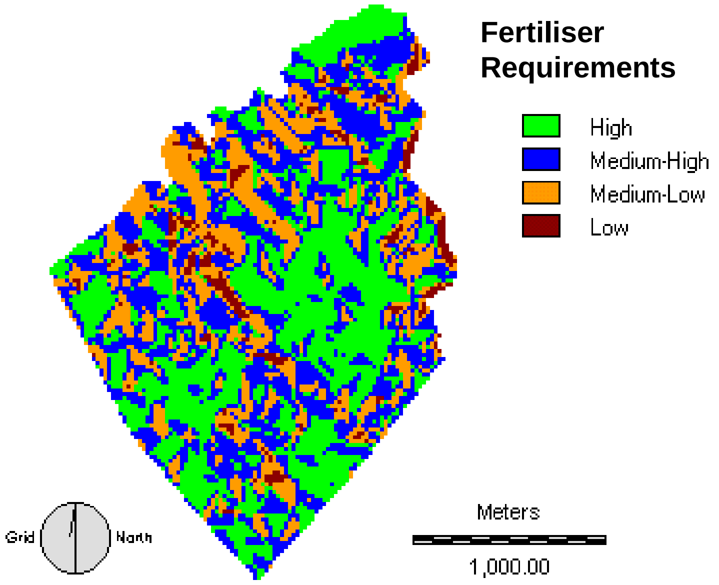

Figure 5.3.1 A digital fertiliser requirement map for hill country ready to control the differential application of fertiliser from a GPS/GIS smart topdressing plane

In hill country this applies particularly to identifying:

1. gullies and streams
2. different slopes and aspects that have marked differences in the potential to grow pasture and therefore have marked differences in the efficiency of nutrient use.
3. steeper areas retired in forestry or native bush.

The differential fertiliser application map can be used to manually avoid fertiliser application to sensitive or low productivity areas. However, because the differential fertiliser application map is held in a digital geographic information system it can eventually be used to generate computer assistance to direct a topdressing plane’s flight path and to automatically control its fertiliser hopper door (application rate). This technology, which is currently under development, has the advantage of not only being able to better target nutrient applications, but also it will provide detailed fertiliser application rate maps for record keeping.

Filtration of surface runoff

Vegetated riparian buffers have been promoted as filters of pollutants carried in surface runoff. Several studies have demonstrated the effectiveness of such a strategy in mitigating diffuse runoff loads from cropland in the United States (Peterjohn and Correll, 1984; Cooper et al., 1987) and from grazed pastures in New Zealand (Smith, 1989). Pollutant removal has been attributed to processes of infiltration, deposition, filtration, adsorption, and absorption. These processes can operate synergistically: for example, infiltration of overland flow reduces runoff velocities, thereby encouraging deposition, and also increases soil-water contact, thereby increasing the opportunity for adsorption. The study of Smith (1989) takes on special significance because it showed that even a narrow riparian filter strip was still effective in reducing edge-of-field losses within a 'typical' New Zealand hill country landscape where runoff velocities and volumes were high due to steep slopes and long hillslope lengths.

The subsequent work of Cooper et al. (1992) showed that a physically-based diffuse source pollution model (called CREAMS) was able to predict the effect of the buffer strip. Given their predictive ability, such models have the potential to guide the design of best management practices (BMP's) for protecting receiving waters from excess nutrient load and this is exactly what they have been used for overseas. If CREAMS is used to predict the effects of alternative riparian filters on the hillslope studied by Smith (1989) and Cooper et al. (1992), simulations predict that for optimum performance the riparian strip needs to be 10 metres wide (but greater width is redundant) and the strip needs to maintain a dense grass sward, at least during winter and spring when the major runoff events occur.

Riparian zone removal of nitrate from shallow groundwaters

Under stable flow conditions, the major transfer of N from land to surface water occurs as nitrate-N via groundwater. Riparian removal of this subsurface nitrate is now a widely-recognized phenomenon, having been observed in a diversity of landscapes around the world. New Zealand research on riparian nitrate removal is extensive (Howard-Williams et al., 1986; Cooper, 1990; Schipper et al., 1991; Cooper, 1993; Schipper et al., 1993). The general consensus from the research is that microbial denitrification to gaseous forms of nitrogen is the predominant nitrate removal process in riparian zones. In addition to nitrate, denitrification requires anoxia and a supply of microbially-assimilable carbon. The position that riparian zones occupy in the landscape favours the development of these conditions and riparian environments variously described as floodplains, bottomlands, wetlands, swamps, seeps and muck soils have all been shown to have a high capacity for denitrification compared to their terrestrial and aquatic neighbours on either side (Schipper et al., 1991).

The New Zealand experience is that nitrate removal efficiencies in such riparian environments exceed 90%, providing that the incoming nitrate has sufficient residence time within the riparian zone (Schipper et al., 1991; Cooper, 1990 and 1993; Schipper et al., 1993). Such high removal efficiencies can result in extremely low nitrate concentrations leaving the riparian zone and entering adjacent watercourses. Practices that reduce residence times, such as artificial drainage, can be expected to drastically reduce nitrate removal efficiencies (Cooper, 1990 and 1993; Haycock and Pinay, 1993) and, from a water quality perspective, should be avoided.

Integrated catchment effects

The majority of studies have focused on a single riparian process and/or have been conducted at the plot-scale. From a water management perspective, it is the integrated effects operating at the catchment scale that are important. There are surprisingly few of these reported in the literature. The one well-documented New Zealand example is the large-scale riparian management scheme that was implemented along the streams draining into Lakes Rotorua and Rotoiti with the aim of reducing sediment and nutrient inputs into the lakes. Eighty percent of the stream margins draining pastoral land were retired from grazing and planted in a variety of trees and shrubs. Both before and after riparian management one of these streams (the Ngongotaha, catchment area 7500 ha) was intensively monitored for sediment and nutrient loads (Cooper et al., 1991).

Monitoring has indicated a significant reduction in the loads of most pollutants in response to riparian management. Field observations suggested that these reductions were due to both deposition of particulates carried in surface runoff from upslope and a reduction in bank erosion. The increase in nitrate load was unexpected and could have been the result of either a long-term increase in groundwater nitrate concentrations due to pastoral development of the catchment or shading by riparian plantings reducing the in-stream uptake of nutrients allowing more to reach the measurement point.

In a follow-up study, Cooper and Bottcher (1993) found that a catchment-scale simulation model they developed was able to predict the observed nutrient loads both before and after implementation of the riparian retirement scheme. Briefly, this simulation model divides the catchment into grid cells and uses information on topography, land use, and soil type to make predictions of nutrient loss from each cell before routing these through riparian zones and the streams to the catchment outlet. This spatial component allows targeting of control measures to those 'cells' where nutrient loss is predicted to be high. Such targeting is an important aid to managers seeking to optimize the cost-efficiency of any catchment-wide management plan.

With the passage of the Resource Management Act, Regional Councils are now charged with giving effect to its purpose of promoting the sustainable management of natural and physical resources and resource users must avoid, remedy, or mitigate any adverse effects of activities on the environment. The means by which effect is given to the RMA is via Regional Policy Statements and, more specifically, catchment management plans. In the agricultural sector there is now a need to recognise the implications of the RMA and, where necessary, adjust farm practices so that environmental outcomes are achieved without compromising economic return. Targeted riparian management seems to offer much because not only does it have the potential to lower nutrient loadings but can also buffer streams against the many other effects of agricultural development (Quinn et al., 1993).

Form and Timing of fertiliser application

As P is strongly adsorbed by the soil, the timely application of fertiliser is more important for N than P. The obvious exception to this rule is when heavy rain follows the addition of P fertiliser and there is increased opportunity for particulate P movement. When applying N, consideration needs to be given to the likelihood of drainage and the ability of the crop to capture the applied N. The worst case scenario would be to apply N prior to a very wet and dull period. In this case, excess rainfall would induce drainage, and as the plant is growing only slowly, most of the applied N would be vulnerable to leaching. The abilities to anticipate plant N requirements and weather will be important components of management skills in the future.

Tillage

No-tillage has been shown to conserve physical and biological attributes of the soil resource but what are its likely effects on the loss of soil nutrients? As stated above, most of the movement of P from tilled systems is as particulate P. As no-tillage reduces runoff and soil erosion it will also decrease the movement of particulate P, although the accumulation of nutrients in the surface soil may increase the concentration of dissolved P exiting in surface runoff from no-tilled areas. Two other features of no-tillage – the increase in permeability and by-pass flow and the retention of crop stubble - do not auger so well for its impact on water quality. Macropores tend to be more continuous and persistent under no-till than conventional cultivation and so there is the risk of greater N leaching under this regime. (This notwithstanding, some researchers have measured less N leaching under no-till than conventional cultivation, presumably due to lower mineralisation rates under no-till and more of the N being retained within aggregates). Likewise, potential immobilisation of N by crop residues may lead to increased rates of N application under no-till. Banding fertiliser application, as opposed to broadcasting, is one way to ensure that N is placed next to plant roots where it is needed. Banded P placement will lessen the risk of P movement in runoff. There have also been very significant responses in crop growth to banded P placement – not surprising given the immobility of P in soil.

Nitrogen leaching from land that is left fallow is likely to be substantial as there is no plant uptake. Autumn sown crops play an important role in reducing movement of nutrients to water by converting nutrients into biomass and minimising soil erosion.

Effluent management

As was discussed above, agriculture effluents are a major threat to water quality. Land treatment is a partial answer. However, systems are required which will help overcome two major problems: the coincidence of an irrigation event and wet soils or rainfall, and the accumulation of nutrients in the soil profile which increases the risk of nutrient movement to surface and ground waters.

Farmers need guidelines to help them manage problems associated with land treatment of effluent. At Massey University, a comprehensive management package called 'deferred irrigation' has been developed to give land managers the information required to overcome these difficulties. A novel aspect of the system is the linking of soil nutrient and water balances in the design and management of the system. Deferred irrigation involves storing effluent in the two-pond treatment system (as found on most farms) and then applying it strategically to land from the aerobic pond. Effluent irrigation is scheduled to coincide with appropriate soil water deficits thereby avoiding surface runoff of effluent and minimising the direct drainage of effluent. Ease of management is achieved by deferring effluent application to four events throughout the lactation season. The use of a soil water balance to schedule irrigation is illustrated in Figure 5.3.2.

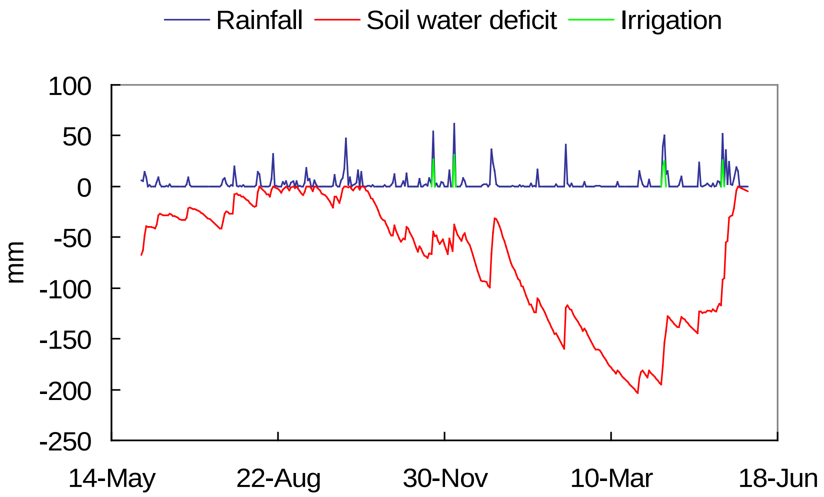

Figure 5.3.2 A soil water balance illustrating how the irrigation of effluent was scheduled at a deficit which precluded drainage and runoff.

In deferred effluent irrigation systems, conserved pasture can be harvested, as hay or silage, from the treatment areas to reduce the accumulation of nutrients in the soil-plant system, thereby, reducing the potential for nutrients to be lost in drainage water or runoff (Table 5.3.1).

Table 5.3.1 The quantities of nutrients applied in effluent compared to the amounts harvested in a crop of baleage.

	Nutrient					
	N	P	K	S	Ca	Mg
Nutrients applied in effluent (kg/100 mm/ha)	180	25	290	2	21	13
Nutrients removed in baleage (kg/5000 kg DM/ha)	100	15	90	12	20	10

Test Your Knowledge

1. How can digital mapping technology be used to reduce nutrient loss to the aquatic environment?

2. Outline the farm management practices which may reduce N and P loss to the surrounding environment?

3. Describe 'deferred irrigation' and how this approach can reduce nutrient loss following effluent application?

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