

## A review of nitrogen and phosphorus export to waterways: context for catchment modelling

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**Abstract.** This paper reviews knowledge of nitrogen and phosphorus generation from land use and export to waterways, including studies relevant to Australia. It provides a link between current and future modelling requirements, and the context for incorporation of this knowledge into catchment models for use by catchment managers. Selected catchment models used by catchment managers are reviewed, and factors limiting their application are addressed. The review highlights the importance of dissolved N and P for overland flow and groundwater pathways, for sheep, beef and dairy grazing land use. Consequently, the effectiveness of riparian buffers to remove N and P may not be adequate. Consideration of the effects of rainfall and hydrology, dissolved P and N losses from pastures and event-based catchment-scale loads are therefore important factors that should be incorporated into catchment models. The review shows that it is likely that nutrient losses under Australian dairying conditions have many similarities to worldwide studies. Catchment models need to represent the importance of event-based loads, intensively farmed land use, management and forms of nutrients. Otherwise there is a likelihood of either underestimating nutrient losses, or potentially overestimating the effectiveness of riparian buffers.

**Extra keywords:** conceptual model, export coefficient, nutrient export, overland flow.

### Introduction

Nitrogen (N) and phosphorus (P) are important nutrient inputs required to maintain the productivity of agroecosystems. However, the mobilisation of both nutrients can have detrimental effects on waterways. Eutrophication in surface waters (e.g. rivers, dams, lakes and estuaries) in combination with other physico-chemical factors (e.g. increased water inflow from rainfall events and stratification) may lead to algal blooms (Robson and Hamilton 2003; Byun *et al.* 2005), resulting in de-oxygenation, reduced aesthetics and increased toxicity, and the loss of, and changes in, species diversity. Eutrophication can also detrimentally affect water for drinking and recreational use (Di and Cameron 2000; Davis and Koop 2001; McDowell *et al.* 2004). Degradation of surface water quality by algal blooms is of significant environmental and economic importance (Gabric and Bell 1993; Webster *et al.* 2001). Yet, surprisingly, there have been relatively few studies on N and P export to waterways in Australia (Marston *et al.* 1995; Young *et al.* 1996), particularly in the valuable

farming region of south-eastern Australia. As a result, reference to northern hemisphere data and assessment techniques is usually necessary (Young *et al.* 1996).

The loss of nutrients from land can be considered to originate from both diffuse and point sources. In agriculturally dominated catchments, diffuse sources usually dominate inputs of N and P into rivers (Davis and Koop 2001; Heathwaite 2003; McDowell *et al.* 2004). For example, the bulk of P carried by rivers in inland Australia is likely to be derived from P sourced from gully and bank erosion (Davis *et al.* 1998b; Wallbrink *et al.* 2003). In contrast, tropical catchment nutrient exports from intensively farmed land have a higher percentage of P in a soluble form, often attributable to fertiliser sources (Davis and Koop 2001; Hunter and Armour 2001).

In Australia, the effect of N on water quality has generally received less attention than P. However, in many freshwater and estuarine systems, inorganic nitrogen, rather than phosphate, can be the limiting nutrient (Ford and Bormans 2000).

Nitrogen supply may therefore limit algal biomass. A low N : P ratio is also associated with increased risk of dominance of cyanobacteria, affecting water quality (Ford and Bormans 2000). Much of the N research in Australia has focussed on estuarine environments rather than rivers and lakes because they often have major population centres surrounding them (Hart and Grace 2000). However, recent research has begun to quantify nitrogen (notably nitrate) fluxes; for example, in riparian zones (Rassam *et al.* 2003), agricultural pasture (Pakrou and Dillon 2000; Mundy *et al.* 2003; Eckard *et al.* 2004) and surface and groundwater (Hunter 2000).

Many northern hemisphere nutrient export studies were suggested by Young *et al.* (1996) to be inappropriate for use under Australian conditions. For example, Harris (2001) indicated that N export from Australian catchments was similar to undisturbed temperate catchments, but less similar than undisturbed tropical catchments. In contrast, N and P exports for disturbed Australian catchments were much lower than for disturbed northern hemisphere catchments (Harris 2001). This was suggested to be a result of generally lower atmospheric deposition (Brodie and Mitchell 2005; Davis and Koop 2006), fertiliser input and population densities than in the northern hemisphere (Harris 2001).

Australia also exhibits a climate that is highly variable with periods of drought and occasional large flood events, often responsible for transporting high sediment and nutrient loads (Mitchell *et al.* 1997; Croke and Jakeman 2001; Webster and Harris 2004). The variability of rainfall and stream discharge in high rainfall areas of Australia is much greater than many parts of the world (Kuhnel *et al.* 1990; Davis and Koop 2006). Similarly, the specific water yield of inland Australian catchments is low by world standards, although coastal river discharges are closer to global averages (Wasson *et al.* 1996). Young *et al.* (1996) suggested that the use of annual average export values in Australia is difficult because the high interannual variability of rainfall makes an 'average' year infrequent, although for long-term planning such values are useful. In addition, Australia generally has poorer quality soils, and more highly weathered landscapes (Wasson *et al.* 1996; Davis and Koop 2006) and less intensive farming systems than the northern hemisphere (Harris 2001). However, this general statement has the potential to obscure the realities in intensively farmed areas and is considered later in our review.

Due to the complex nature of agricultural systems, modelling approaches are commonly used to help understand the long-term nutrient export status of catchments, and to predict changes in land use and management. Without the use of models, correctly interpreting water-quality trends based on limited measurements, often obtained at high cost, can be difficult because of high temporal variability (Molloy and Ellis 2002). Models developed in Australia range from simple nutrient export rate models, e.g. the Catchment Management

Support System (Davis and Farley 1997), through to more complex integrated catchment models, e.g. CatchMODS (Newham *et al.* 2004b).

The objectives of this paper are to (1) review knowledge on N and P export generation from diffuse sources with particular reference to Australian catchments, (2) review catchment-scale diffuse source nutrient modelling for catchment management and (3) suggest where future research development should be targeted.

### Overview of nutrient sources and transport processes

Nitrogen and phosphorus are available from many diffuse sources and are subject to a variety of transport processes to waterways. This section presents an overview of these sources and pathways. Many aspects of the processes and pathways of diffuse N and P transfer from soil have been studied in detail and documented elsewhere (e.g. Haygarth *et al.* 1998; Haygarth and Jarvis 1999; Ledgard *et al.* 1999; McDowell *et al.* 2004).

#### Nitrogen

Soil nitrogen inputs are available from nitrogen fixation via legume pastures, plant residues and fertiliser N (Pakrou and Dillon 2000; Cameron *et al.* 2002). However, leaching of N frequently occurs via high N concentrations from grazing-animal urine patches, rather than from direct fertiliser losses (Silva *et al.* 1999; Di and Cameron 2000; Cameron *et al.* 2002; Monaghan *et al.* 2002). Soil nitrogen is frequently present in soil organic matter and is not available for plant uptake or leaching unless mineralised to nitrate or ammonium. In years when drainage volumes are low, pastoral systems can utilise N fertiliser without impact on waterways. However, nitrate may accumulate in soil in low drainage years and be displaced in subsequently higher drainage years, contributing to greater N losses (Tyson *et al.* 1997; Ridley *et al.* 2001; Eckard *et al.* 2004). The cumulative nitrate load in a waterway is important, and depends on the cumulative intensity of grazing and fertiliser use in a catchment (Eckard *et al.* 2004).

#### Phosphorus

Losses of P to waterways can be via natural soil P (termed non-incident) and increased soil fertility via soil additions of fertiliser, manure or dung (incident). These have been reviewed and studied widely (e.g. Cameron *et al.* 2002; McDowell *et al.* 2004). However, it is the combination of source, release and transport factors that will determine overall P loss to waterways (Heathwaite 2003; McDowell *et al.* 2004).

Phosphorus has several forms. Total P (TP) includes particulate P (PP) and dissolved reactive P (DRP). Dissolved reactive P, or filterable P, is commonly measured by passing through a 45- $\mu\text{m}$  filter, and therefore may include P attached to fine soil colloids that pass through the filter (Nash *et al.*

2002). To avoid confusion, Haygarth and Sharpley (2000) suggested 'filterable P' as an alternative for the established equivalent terms DRP and orthophosphate. In terms of the potential for eutrophication, DRP is considered fully available to aquatic plants, whereas only a proportion of PP is available (Gabric and Bell 1993; McDowell *et al.* 2004; Brodie and Mitchell 2005). Generally, the use of DRP is recommended for measuring the nutrient status of rivers, whereas TP is preferred for lakes, as much particulate P is present in algae where it can be solubilised and made available for further algal growth (McDowell *et al.* 2004).

#### Pathways of N and P transport

Pathways for N and P movement have been discussed in several reviews and studies including Haygarth and Jarvis (1999), Cameron *et al.* (2002), Nash *et al.* (2002) and McDowell *et al.* (2004). Field-scale hydrology affecting nutrient movement can be divided into (1) surface or overland flow and (2) subsurface pathways. Nitrogen, for example, can be transported to groundwater via soil macropores (Rassam *et al.* 2003), leaching (Eckard *et al.* 2004; Monaghan *et al.* 2005), or overland flow (Pakrou and Dillon 1995, 2000; Mundy *et al.* 2003). The connectivity of the land to the waterway is also important for P loss (Sharpley *et al.* 2003; McDowell *et al.* 2004).

Historically, it has been assumed that P is not often transported through soil, but recent work refutes this assumption (Stevens *et al.* 1999; Monaghan *et al.* 2002; Toor *et al.* 2004). Phosphorus can be transported through soil often via soil macropores (Cox *et al.* 2000; Haygarth and Sharpley 2000). On a catchment scale, the subsurface pathway provides nutrients and stream baseflow when storm events are absent (Cosser 1989; McDowell *et al.* 2004).

Many areas of grazing and cropping land in Australia have soils that are subject to structural breakdown when wet (Greene and Hairsine 2004), contributing to gully erosion. Although rates of gully erosion and, to a lesser extent, stream bank erosion have declined over recent decades, these processes still continue to be a dominant source of P in many areas (Caitcheon *et al.* 1999).

#### Studies of nutrient generation in Australian catchments

Land use and management policies were captured in the Catchment Management Support System (CMSS) model to assist catchment managers to assess likely impacts of land-use nutrient generation and management on the long-term loads of catchments (Davis and Farley 1997). Further details of the model are discussed in a later section. Total N and P generation rates from 16 Australian and 32 worldwide studies thought to be appropriate for Australian conditions were collated in the Nutrient Data Book (Marston *et al.* 1995). The Nutrient Data Book was designed to assist CMSS users to estimate annual average nutrient generation rates for use in that model.

Studies of land-use types in the Nutrient Data Book are summarised in Table 1. Land use can be used as a simple predictor of nutrient loads, but that conclusion is based on limited literature (Young *et al.* 1996). Further, the greatest number of studies was obtained from urban land use, followed by improved pasture. Although limited adjustments can be made for differences between catchments, it is much more difficult to account for the effects of climate, drainage and gully density, soil fertility and management.

We now review Australian studies published after 1996 quantifying N and P losses under pastoral agriculture and forest. These studies are discussed in relation to worldwide research. We consider the soluble components of N and P exports and their importance to waterway health, and discuss research that should be incorporated into catchment models.

#### Sheep and cattle grazing nutrient export

Recent studies show the importance of soil leaching and dissolved nutrients in soil drainage and overland flow. Although losses of nitrogen from soil drainage and overland flow from texture contrast (duplex) soils are generally low, nitrate losses of 13–33 kg N ha<sup>-1</sup> have been reported during a wet year on these soils (Ridley *et al.* 2001). This is significantly greater than previous studies. Leaching losses of nitrogen varied from 9 to 15 kg N ha<sup>-1</sup> year<sup>-1</sup> when drainage

**Table 1. Overview of land uses summarised within the Nutrient Data Book (Marston *et al.* 1995)**

TP, Total phosphorus; TN, total nitrogen

Land use	Number of studies in Australia	Generation rate for south-east Australia (kg ha <sup>-1</sup> year <sup>-1</sup> )	
		TP	TN
Mixed farming or rural	5	0.2, 0.3, 0.57	4.5
Forest/scrub	6	0.03, 0.05 <sup>A</sup> , 0.07 <sup>A</sup>	0.9
Pasture grazing	8	0.12, 0.25, 0.35, 0.9, 1.6	0.62, 0.6–1, 4.6
Pasture native	2	0.1	1.5
Horticulture	2	2.7, 6.4	6.4, 20.1, 26
Urban	8	0.39, 0.4, 0.91, 1.1, 3.6 <sup>B</sup>	3.2, 3.7, 5.3, 22.8 <sup>B</sup>

<sup>A</sup>Plantation forest (*Pinus* species).

<sup>B</sup>Commercial/industrial.

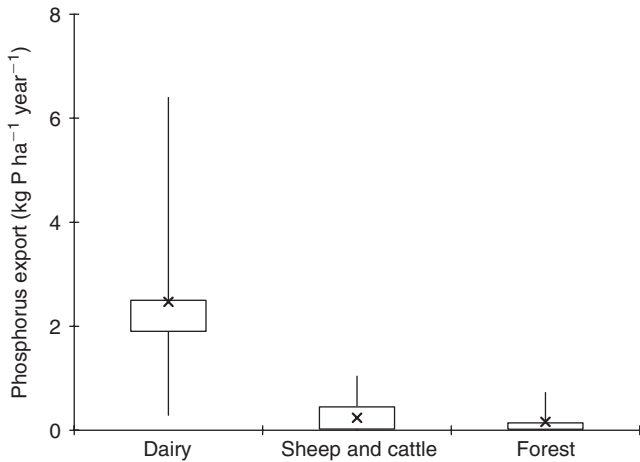
occurred under sheep-grazed pasture with no incidental N applied (Ridley *et al.* 2001). However, these pastures were sown 2 years before measurements, which may have contributed to greater mineralisation, hence drainage nitrate content. Although management, soils and drainage vary, studies worldwide show broadly similar results. For example, over 8 years, mean nitrate leaching losses of  $12.9 \text{ kg N ha}^{-1} \text{ year}^{-1}$  were recorded in grass/clover pastures with no incidental N application (Tyson *et al.* 1997). However, mean leaching losses increased to  $50 \text{ kg N ha}^{-1} \text{ year}^{-1}$  in pastures fertilised with  $200 \text{ kg N ha}^{-1} \text{ year}^{-1}$  and ranging from 29 to  $72 \text{ kg N ha}^{-1} \text{ year}^{-1}$  (Tyson *et al.* 1997). Although leaching losses at the same UK site increased greatly with  $400 \text{ kg N ha}^{-1} \text{ year}^{-1}$  applied (Scholefield *et al.* 1993), such application rates are rare under common practice beef or even dairy grazed pasture in Australia and New Zealand. Farm management practices such as grazing of forage crops, cultivation and manure or fertiliser applications are also important determinants of N leaching, so it can be difficult to compare studies directly (Silva *et al.* 1999; Allingham *et al.* 2002). Nutrient export rates and N and P concentrations for these studies are summarised in Table 2.

Catchment groundwater discharge, even in low rainfall conditions, is an important pathway for N and P loss (Cox and Ashley 2000). Similarly, environmentally significant concentrations of N and P are moved in both drainage and overland flow (Stevens *et al.* 1999) (Table 2). The importance of dissolved P to nutrient exports in less intensive sheep-grazed dry land pastoral systems has also been reported by Johnston *et al.* (2003). The soluble to TP ratio in runoff plots was higher (81%) in improved pasture, which received more frequent fertiliser applications and heavier grazing, than in native pasture (62%; Johnston *et al.* 2003). In contrast to runoff plots, the ratios were less when measured from paddock catchment outlets. Exports of P from this and a range of other locations ( $0.01\text{--}0.56 \text{ kg P ha}^{-1} \text{ year}^{-1}$ ) are summarised elsewhere for a range of conditions (McCaskill *et al.* 2003). Although P export rates in these studies are generally less than some UK exports reported in Table 2, the exports in the study of Haygarth and Jarvis (1997) may have been high owing to fertiliser, cattle excreta and soil poaching, although the effects of these were not presented. These studies, and those presented in Table 2, highlight the importance of dissolved nutrients when flow is primarily from groundwater discharge. Consequently, rainfall and hydrology and dissolved P and N losses from sheep- and beef-grazed pastures are important factors that should be incorporated into catchment models.

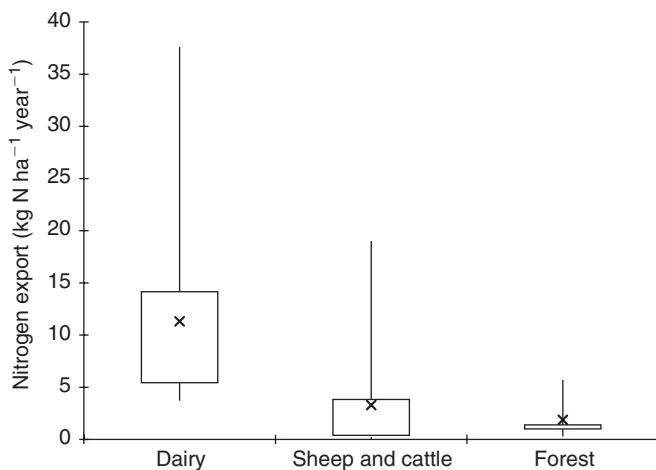
Nutrient export rates for the Australian studies in this review are shown in Fig. 1 and Fig. 2. Not unexpectedly, nutrient exports from sheep and beef grazing are less than dairy grazing. However, this review shows that some nutrient export rates under sheep and beef grazing are much greater than for the previous studies reported in Table 1.

**Table 2. Annual average total and dissolved phosphorus and nitrogen generation rates and concentrations from dryland sheep and cattle grazing field studies**  
OF, Overland flow; TP, total phosphorus; TN, total nitrogen

Hydrological process	Region	TP ( $\text{kg P ha}^{-1} \text{ year}^{-1}$ )	Dissolved P ( $\text{kg P ha}^{-1} \text{ year}^{-1}$ )	TP ( $\text{mg L}^{-1}$ )	Dissolved P ( $\text{mg L}^{-1}$ )	TN ( $\text{kg N ha}^{-1} \text{ year}^{-1}$ )	NO <sub>3</sub> ( $\text{kg NO}_3\text{-N ha}^{-1} \text{ year}^{-1}$ )	TN ( $\text{mg L}^{-1}$ )	NO <sub>3</sub> ( $\text{mg L}^{-1}$ )	Size	Reference
Leaching	Vic, Australia									Plots	Ridley <i>et al.</i> (2001)
Leaching	Vic, Australia	0.09–0.45								3–17 ha	Ridley <i>et al.</i> (2003)
Leaching	England									0.4 ha plots	Cuttle <i>et al.</i> (1998)
Leaching	England									0.4 ha plots	Cuttle <i>et al.</i> (1998)
Leaching	England										Tyson <i>et al.</i> (1997)
Leaching	England										Tyson <i>et al.</i> (1997)
Drainage	SA, Australia	To 0.14	70%	0.1–0.5						Plots	Cox and Pitman (2001)
through flow											
OF & through	SA, Australia	To ~0.20	8–50%	0.13–1.02	0.01–0.53					2–4 ha	Stevens <i>et al.</i> (1999)
flow											
OF	Australia	0.01–0.56		Various						Various sites	McCaskill <i>et al.</i> (2003)
OF	Vic, Australia									Plots	Ridley <i>et al.</i> (2001)
OF	Vic, Australia	0.04–0.47		0.12–0.92						3–17 ha	Ridley <i>et al.</i> (2003)
OF	SA, Australia	To 0.07	35%	0.1–0.5						Plots	Cox and Pitman (2001)
OF	Ireland		0.69–4.7 (77%)							Plots	Kurz <i>et al.</i> (2005)
OF	England	3	32%	0.12	0.09–1.0					1 ha plots	Haygarth and Jarvis (1997)
interflow					0.04						
Stream	SA, Australia	1.04	0.14	0.35	0.048	9.0	4.1	1.68	1.38	3 km <sup>2</sup>	Nelson <i>et al.</i> (1996)
Gully stream	SA, Australia	(0.005–0.007)	100%				0.002			200 ha	Cox and Ashley (2000)
Stream	NSW, Australia	0.8		0.28 EMC		7.0		2.1 EMC		4 ha	Hollinger and Cornish (2002)



**Fig. 1.** Phosphorus export rates ( $\text{kg P ha}^{-1} \text{ year}^{-1}$ ) for Australian field studies. The box is between lower and upper quartile, whiskers are maximum and minimum, x is the mean.



**Fig. 2.** Nitrogen export rates ( $\text{kg N ha}^{-1} \text{ year}^{-1}$ ) for Australian field studies. The box is between lower and upper quartile, whiskers are maximum and minimum, x is the mean.

**Dairy land-use nutrient export for hydrological pathways**

Numerous worldwide studies show dairying to have a generally high generation of N and P, but it is only recently that such work has been carried out in Australia. Nutrient generation rates from dairying are summarised in Tables 3–7, with Tables 3–5 showing export via the overland flow pathway. Overland flow is an important pathway for P loss under dairying, with single storm events often responsible for high loads. For example, 69% of annual TP loss was reported as lost in a single storm event by Nash and Murdoch (1997), whereas Fleming and Cox (2001) showed 98% of TP loss was in overland flow during a 3-year period. However, observed losses vary considerably depending on rainfall, with most losses

**Table 3. Annual average total and dissolved phosphorus generation rates and concentrations from overland flow from dairy field studies**

Range is given where mean is not available. OF, Overland flow; TP, total phosphorus; TN, total nitrogen

Hydrological pathway	Region	TP ( $\text{kg P ha}^{-1} \text{ year}^{-1}$ )	Dissolved P ( $\text{kg P ha}^{-1} \text{ year}^{-1}$ )	TP ( $\text{mg L}^{-1}$ )	Dissolved P ( $\text{mg L}^{-1}$ )	Annual P fertiliser ( $\text{kg P ha}^{-1}$ )	Comment	Size	Reference
OF	Vic, Australia	3.43	3.2 kg 93%	5.2	93%	60–84		3.6 ha	Nash and Murdoch (1997)
OF & interflow	SA, Australia	To 2.4	50–60%	0.3–2.1	0.1–1.4	15	92% TP lost in OF	2.2–2.6 ha	Fleming and Cox (1998)
OF & interflow	SA, Australia	To 2.3	45%	Up to ~3		15	98% TP lost in OF	2.2–2.6 ha	Fleming and Cox (2001)
OF	SA, Australia	2.43	0.95 (39%)			15		2.2 ha	Fleming <i>et al.</i> (2001)
OF	SA, Australia	0.29	0.14 (48%)			15		2.6 ha	Fleming <i>et al.</i> (2001)
OF	NSW, Australia	1.9	56%	0.95		0		4 ha	Cornish <i>et al.</i> (2002)
OF	NSW, Australia	2.5	59%	0.66		0		140 ha	Cornish <i>et al.</i> (2002)
OF on drained plots	New Zealand	0.031	0.009			55–75 <sup>A</sup>	0–400 $\text{kg N ha}^{-1}$	0.09 ha	Smith and Monaghan (2003)
OF on undrained plots	New Zealand	0.23	0.09			55–75 <sup>A</sup>	0–400 $\text{kg N ha}^{-1}$	0.05 ha	Smith and Monaghan (2003)

<sup>A</sup>75 kg year<sup>-1</sup>; 55 kg P years 2–3.

**Table 4. Annual average total and dissolved phosphorus generation rates and concentrations from overland flow from flood irrigation on dairy land use field studies**

Range is given where mean is not available. TP, Total phosphorus

Region	TP (kg P ha <sup>-1</sup> year <sup>-1</sup> )	Dissolved P (kg P ha <sup>-1</sup> year <sup>-1</sup> )	TP (mg L <sup>-1</sup> )	Dissolved P (mg L <sup>-1</sup> )	Cows ha <sup>-1</sup>	Soil Olsen P (mg kg <sup>-1</sup> )	Annual P fertiliser (kg P ha <sup>-1</sup> )	Comment	Size	Reference
Vic, Australia	3.6–2.3	80–95%	3–11	3–9	3.4		0, 20, or 29	Flood irrigation re-use	Paddock	Barlow <i>et al.</i> (2005)
Vic, Australia	2.5–14	82–91%	2.2–6.2	2–5.1	3.4		0, 20, or 29	Flood irrigation re-use	Farm section	Barlow <i>et al.</i> (2005)
Vic, Australia	0.24–0.46	0.12–0.25	1.31–2.19	0.59–1.29	Un-grazed	33	45	Ungrazed 1 flood irrigation only	Paddock	Mundy <i>et al.</i> (2003)
Vic, Australia	0.84–1.08					33	45	4 flood irrigations, grazed once	Paddock	Mundy <i>et al.</i> (2003)
Vic, Australia	6–17 <sup>A</sup>		1.7–2.8 (3.3–20 <sup>A</sup> )	1.1–2.1	1.8–3.8	8–46	16–81	Volume dependent	7.6–11.6 ha	Nexhip and Austin (1998)
New Zealand	3.4	80%		0.6		45	~50	7 flood irrigation events, each grazed	0.8 ha	Carey <i>et al.</i> (2004)

<sup>A</sup>Highest losses followed fertiliser application.**Table 5. Annual average total and dissolved nitrogen generation rates and concentrations from overland flow and streams for dairy land use field studies**

OF, Overland flow; TP, total phosphorus; TN, total nitrogen

Process	Region	TN (kg N ha <sup>-1</sup> year <sup>-1</sup> )	NO <sub>3</sub> (kg NO <sub>3</sub> -N ha <sup>-1</sup> year <sup>-1</sup> )	TN (mg L <sup>-1</sup> )	Comment	Size	Reference
OF	New Zealand	0.046 <sup>A</sup>	0.023		Drained plots 0 kg N ha <sup>-1</sup>	0.09 ha	Smith and Monaghan (2003)
OF	New Zealand	0.084 <sup>A</sup>	0.036		Drained plots 400 kg N ha <sup>-1</sup>	0.09 ha	Smith and Monaghan (2003)
OF	New Zealand	0.333 <sup>A</sup>	0.19		Un-drained plots 0 kg N ha <sup>-1</sup>	0.05 ha	Smith and Monaghan (2003)
OF	New Zealand	0.841 <sup>A</sup>	0.48		Un-drained plots 400 kg N ha <sup>-1</sup>	0.05 ha	Smith and Monaghan (2003)
OF	New Zealand	2.0			7 Flood irrigation events, each grazed	0.8 ha	Carey <i>et al.</i> (2004)
OF	Vic, Australia	0.23–0.4 <sup>B</sup>		1.14–1.91	1 Flood irrigation only, un-grazed		Mundy <i>et al.</i> (2003)
OF	Vic, Australia	1.31–2.35 <sup>C</sup>			4 Flood irrigations only, grazed once		Mundy <i>et al.</i> (2003)
OF	Vic, Australia	10–16		2.8–6.2 (3.1–17 <sup>D</sup> )	Flood irrigation, vol dependent	7.6–11.6 ha	Nexhip and Austin (1998)
Stream	SA, Australia	6.6	0.4	2.27	Sand over clay, stream	3 km <sup>2</sup>	Nelson <i>et al.</i> (1996)
Stream	NSW, Australia	4.1, 4			Dryland dairy	4–200 ha	Hollinger and Cornish (2002) and Baginska <i>et al.</i> (1998)
Stream	NSW, Australia	5.8, 6			Irrigated dairy	44 ha	Hollinger and Cornish (2002) and Baginska <i>et al.</i> (1998)
Stream	New Zealand	35.3	30.7		Dairy catchment	15 km <sup>2</sup>	Wilcock <i>et al.</i> (1999)

<sup>A</sup>Nitrate + ammonium only.<sup>B</sup>1 flood irrigation only.<sup>C</sup>Total inorganic N.<sup>D</sup>Following fertiliser application.

**Table 6. Annual average nitrate leaching generation rates and drainage water concentrations for fertiliser applications from dairy pasture field studies**

Region	NO <sub>3</sub> (kg NO <sub>3</sub> -N ha <sup>-1</sup> year <sup>-1</sup> )	NO <sub>3</sub> (mg L <sup>-1</sup> )	N fertiliser application (kg N ha <sup>-1</sup> year <sup>-1</sup> )	Size	Reference
Vic, Australia	3.7–14.6	1.9–4.7	0	0.062 ha	Eckard <i>et al.</i> (2004)
Vic, Australia	6.2–22	3–5.6	200 urea	0.062 ha	Eckard <i>et al.</i> (2004)
Vic, Australia	4.3–37.6	2.3–11.4	200 ammonium nitrate	0.062 ha	Eckard <i>et al.</i> (2004)
New Zealand	40	7	0	Farmlet	Ledgard <i>et al.</i> (1998)
New Zealand	81	10	200	Farmlet	Ledgard <i>et al.</i> (1998)
New Zealand	25 (TN 26.5)	6.9 (TN 7.3)	0	0.09 ha	Monaghan <i>et al.</i> (2002)
New Zealand	30	7.9	0	0.09 ha	Monaghan <i>et al.</i> (2005)
New Zealand	46	12.3	200	0.09 ha	Monaghan <i>et al.</i> (2005)
New Zealand	56	15	400	0.09 ha	Monaghan <i>et al.</i> (2005)

(up to 2.3 kg P ha<sup>-1</sup>) occurring during the wettest year, whereas losses in dry years were up to ~0.5 kg P ha<sup>-1</sup> (Fleming and Cox 2001). In contrast, the magnitude of N and P loss was considerably less under sheep and cattle farming (Figs 1, 2).

Dissolved P has been found to be an important component of TP loss via overland flow (Haygarth and Jarvis 1997; McDowell and Sharpley 2001; Kurz *et al.* 2005). Similarly, Nash and Murdoch (1997) showed that 93% of the total P lost annually in overland flow was in dissolved form. However, most runoff in the study of Nash and Murdoch (1997) occurred when soil was saturated, and the authors suggested that riparian buffers would therefore be unlikely to reduce P losses. Soluble forms of P also dominated dairy runoff in many of the other studies listed in Table 3. For example, Cornish *et al.* (2002) reported a high proportion of dissolved P in runoff, even though there were no regular applications of fertiliser P. Although studies have shown the importance of soil P levels to subsequent P losses (McDowell and Sharpley 2001; Kurz *et al.* 2005), further research linking nutrient movement from paddocks to streams is warranted. Once the agronomic optimum soil P target is reached, further P fertiliser applications are not required. However, these targets are often exceeded in intensively farmed areas, with subsequently greater risk of P loss to waterways (Drewry *et al.* 2003; Monaghan *et al.* 2003; McDowell *et al.* 2004). This is particularly the case for sandy soils with low P retention (Weaver and Reed 1998).

Flood, or border-check irrigated dairying may contribute to large losses of nutrients via overland flow. Losses of P and N with flood irrigation are summarised in Tables 4 and 5. The greatest losses reported occurred when fertiliser had been recently applied, and losses were considerably reduced in subsequent irrigations (Nexhip and Austin 1998). Such studies highlight the importance of implementing best management practices and water re-use schemes (Anon 1998). Worldwide research indicates that leaching is an important pathway for N loss under flood irrigation, but less nitrate may be leached under flood than spray irrigation, possibly

due to greater denitrification in wetter conditions (Di *et al.* 1998).

Leaching through soil is also an important pathway for N losses under dairying. Numerous global studies have shown that nitrate leaching increases with stocking rate or fertiliser application (Tyson *et al.* 1997; Monaghan *et al.* 2005; Table 6). This is thought to be a result of the high N content of urine patches (Silva *et al.* 1999; Di and Cameron 2000). Similarly, field drainage studies have shown 25–40 kg NO<sub>3</sub>-N ha<sup>-1</sup> year<sup>-1</sup> leached without incidental N application, indicating that losses inevitably occur due to natural soil mineralisation and flushing processes (Ledgard *et al.* 1998; Monaghan *et al.* 2002). Nitrate losses increased to 81 and 152 kg N ha<sup>-1</sup> year<sup>-1</sup> under 200 and 400 kg urea N applied ha<sup>-1</sup> year<sup>-1</sup>, respectively (Ledgard *et al.* 1998). Similarly, nitrate losses of 46 and 56 kg N ha<sup>-1</sup> year<sup>-1</sup> were leached under 200 and 400 kg urea N applied ha<sup>-1</sup> year<sup>-1</sup>, respectively (Monaghan *et al.* 2005). In one of the few Victorian dairy field leaching studies, nitrate leaching exports (Table 6) from equivalent N applications (Eckard *et al.* 2004) were generally less than the global studies discussed above. However, the nitrate concentrations for all these studies would be likely to have exceeded guidelines for weed and algal growth (ANZECC 2000). Such studies do not take account of factors important at catchment scale (e.g. dilution and stream processes).

Although N leaching under grazing is an important loss pathway, application of effluent has also been shown to be important. Dairy effluent contributes to less N leaching loss than for equivalent N loadings in grazed systems, because of the predominant organic form of N in effluent (Di *et al.* 1998; Silva *et al.* 1999; Di and Cameron 2000). In contrast, other research shows that organic matter from farmyard manure can build up during many years of application, and therefore contribute to long-term leaching once equilibrium mineralisation is reached (Whitmore and Schroder 1996). Recent research shows elevated P and N losses via macropores from effluent application (Houlbrooke *et al.* 2004; Toor *et al.* 2004). Although studies have examined P sorption of

**Table 7. Annual average total and dissolved phosphorus generation rates and concentrations from leaching and in-stream from dairy field studies**  
Range is given where mean is not available. TP, Total phosphorus

Hydrological pathway	Region	TP (kg P ha <sup>-1</sup> year <sup>-1</sup> )	Dissolved P (kg P ha <sup>-1</sup> year <sup>-1</sup> )	TP (mg L <sup>-1</sup> )	Comment	Size	Reference
Leaching	New Zealand	0.16–0.43	0.05–0.11	0.1–0.2	0–400 kg N ha <sup>-1</sup>	0.09 ha	Monaghan <i>et al.</i> (2005)
Leaching	New Zealand	0.15	0.059 (39%)		0 kg N ha <sup>-1</sup>	0.09 ha	Monaghan <i>et al.</i> (2002)
Stream	SA, Australia	1.11	0.84	0.4	Sand over clay	3 km <sup>2</sup>	Nelson <i>et al.</i> (1996)
Stream	NSW, Australia	1.9–2.5		1.5 EMC (to 5)	Dryland dairy	4 ha	Hollinger and Cornish (2002) and Baginska <i>et al.</i> (1998)
Stream	NSW, Australia	6–6.4		4 EMC	Irrigated dairy	44 ha	Hollinger and Cornish (2002) and Baginska <i>et al.</i> (1998)
Stream	New Zealand	1.16	0.54		Dairy catchment	15 km <sup>2</sup>	Wilcock <i>et al.</i> (1999)

effluent (Holford *et al.* 1997), there have been very few published studies in Australia examining environmental losses of N and P from dairy effluent application. Consequently, further research on nutrient loss under dairy effluent application is required.

Historically, it has been assumed that P does generally not leach through soil but recent work shows this is not always the case (Stevens *et al.* 1999; Monaghan *et al.* 2002). Macropores can transport P (Cox *et al.* 2000; Haygarth and Sharpley 2000), but are subject to seasonal and grazing management changes (Drewry 2006). Macropore flow was found to be an important pathway for P in the study of Kirkby *et al.* (1997) where it was shown that P transport to subsurface horizons was more dependent on hydrology and presence of macropores, than on soil chemistry.

From our discussion of studies presented in this review, it is therefore likely that nutrient losses under dairying in Australian conditions have many similarities to worldwide studies, under similar management. Although rainfall and soil drainage are highly variable, it is only recent research that has studied environmental losses of N and P in Australian dairying. Recent studies under grazing show the importance of the leaching pathway in Australia. However, diffuse nitrogen generation from agriculture is generally not well quantified in Australia.

### Forest nutrient export

There have been few reported studies of nutrient export from entirely forested catchments in Australia. However, several studies have reported catchment nutrient exports from mixed agriculture and forested catchments. In this section, we concentrate on predominantly forested catchments. Nutrient concentrations and losses in tropical catchments and forests have been reviewed elsewhere (Brodie and Mitchell 2005).

Nutrient export rates from forested areas are summarised in Table 8. Studies have included predominantly *Eucalyptus* and *Nothofagus* forest, with a small area of secondary regrowth forest after logging (Campbell 1978). However, as with any nutrient study it is important to consider if storm events were monitored. For example, monthly water samples were collected in the study of Campbell (1978) but it is likely that nutrient export during storm events, particularly sediment-bound P, was underestimated. In contrast, three storm flow events monitored after drought for 3 months yielded 0.01 kg TP ha<sup>-1</sup> and 1 kg total nitrogen (TN) ha<sup>-1</sup> in a catchment mostly unaffected by fire (Chessman 1986; Table 8). In contrast, the storm events monitored after a fire that occurred in 50–98% of catchment area yielded 2–3 times greater TN per unit area, namely 0.07–0.27 kg TP ha<sup>-1</sup> and 2.3–2.8 kg TN ha<sup>-1</sup> (Chessman 1986). Nutrient loss may also be less in plantation than in native forest (Cooper and Thomson 1988), although this study had examined losses before harvesting.



Table 8. Annual average phosphorus and nitrogen generation rates and stream concentrations from forest catchment studies

Range is given where mean is not available. TP, Total phosphorus; TN, total nitrogen

Region	TP (kg P ha <sup>-1</sup> year <sup>-1</sup> )	TN (kg N ha <sup>-1</sup> year <sup>-1</sup> )	NO <sub>3</sub> (kg NO <sub>3</sub> -N ha <sup>-1</sup> year <sup>-1</sup> )	TN (mg L <sup>-1</sup> )	Comment	Size	Reference
NSW, Australia	(0.01–0.07)	(0.2–1.1)		0.25–0.26 <sup>A</sup>	Bushland	25–174 km <sup>2</sup>	Hollinger and Cornish (2002)
Qld, Australia				0.24 <sup>B</sup>	Forest		Bramley and Roth (2002)
NSW, Australia	(0.2–0.73)	(1.4–5.7)			7 subtropical catchments, 50–94% forest	43–17 000 km <sup>2</sup>	Eyre and Pont (2003)
Vic, Australia	0.024				All forest, some regrowth.	13 km <sup>2</sup>	Campbell (1978)
Vic, Australia	0.083				Eucalyptus, Nothofagus		
Vic, Australia	0.01 <sup>C</sup>	1.0 <sup>C</sup>			65% forest (Eucalyptus), some logging, 35% pasture	140 km <sup>2</sup>	Campbell (1978)
New Zealand	0.09	1.31	0.55		Wet sclerophyll forest (7% affected by fire)	720 km <sup>2</sup>	Chessman (1986)
New Zealand	0.12	3.7	2.84		Pine plantation	0.34 km <sup>2</sup>	Cooper and Thomsen (1988)
Americas		5.1 (0.7–8)	2.43		Native podocarp & hardwood	0.28 km <sup>2</sup>	Cooper and Thomsen (1988)
USA			~1–30		Tropical forest	Varied	Lewis <i>et al.</i> (1999)
					Forest (oak, maple)	39–1156 km <sup>2</sup>	Swistock <i>et al.</i> (1997)

<sup>A</sup>Median monthly flow weighted concentration. <sup>B</sup>Median values. <sup>C</sup>Three months only.

These and worldwide studies show generally lower nutrient generation rates from forest in contrast to the pastoral land-use studies discussed in previous sections. In forested catchments, the predominant form of N is particulate and dissolved organic N (Harris 2001; Eyre and Pont 2003; Brodie and Mitchell 2005), whereas after clearing, exports increase and the predominant form of N is dissolved inorganic N (Harris 2001; Brodie and Mitchell 2005). Additional research is suggested to evaluate the forms of P and N from forested areas. Recent research has evaluated overland flow contaminant loss from forest roads (Forsyth *et al.* 2006), tracks and timber harvesting areas (e.g. Croke *et al.* 1999), but a knowledge gap still exists linking contaminant generation to streams from natural and plantation forests.

### Implications for catchment modelling

#### Riparian zones

The review has shown that the forms and quantities of N and P vary with land management and land use. The effectiveness of riparian buffers to remove dissolved N and P may not be adequate (Kirkby *et al.* 1997; Nash and Murdoch 1997; Hunter 2000; McDowell *et al.* 2004). Although riparian vegetation and wetlands provide an opportunity for removal of nutrients, the riparian zone may have a finite lifespan to effectively remove nutrients. Worldwide, there is increasing concern that the effectiveness of riparian buffers to remove P from runoff declines with time. Riparian buffers may potentially become a nutrient source from nutrient accumulation over time (Cooper *et al.* 1995; McDowell *et al.* 2004). McKergow *et al.* (2003) found that in a small agricultural catchment, improved riparian management reduced sediment exports (a likely result of reduced stream bank erosion), but there was little effect on overall N exports, TP concentration and loads, although the soluble P proportion had increased. Although grass riparian strips have been shown to be more effective at filtering sediment than forest buffers (McKergow *et al.* 2004), there has been little detailed research into the effectiveness of N removal by buffer strips in Australian systems (Hart and Grace 2000; Hunter 2000). Given the increasing investment in fencing farm riparian areas, improved simulation of nutrient removal is required. Recent research in this area includes the construction of a model of riparian particulate trapping (Newham *et al.* 2005b). Other considerations include the increase in macropore volume and infiltration after animals are removed from grazed or riparian areas (Drewry 2006).

It is of concern that dissolved nutrients may not be adequately included in catchment models, particularly when evaluating riparian management of grazed pastoral systems. A major gap in knowledge is the capacity to predict different forms of N entering waterways in Australia, as commonly used models predict only TN (Hart and Grace 2000). Our review has also indicated that many agricultural systems export dissolved P and N from both overland

flow and leaching pathways, which should be taken into account in catchment-scale modelling. Other factors affecting the source, transport and subsequent delivery of nutrients, including farm dams, are also important to incorporate into catchment models. Some of these factors are discussed in the later modelling section.

#### *Effect of scale*

The ability of models to predict nutrient loss is often related to the scale of model development and application. Issues associated with scale are therefore briefly discussed in this section. Much of the nutrient export data has been derived from small-scale field trials but these do not always retain the same hydrological pathways at catchment scale (Heathwaite 2003). Few studies have attempted to link transport factors from lysimeter-based experiments to streams, although attempts have been made to incorporate this information into larger-scale nitrogen models in New Zealand (Di and Cameron 2000). At the plot scale, soil and crop type, nutrient cycling and leaching dominate (Quinn 2004), whereas hydrological transport processes dominate at the hillslope scale (Nash *et al.* 2002; Quinn 2002; McDowell *et al.* 2004). At a large catchment scale, key influences include land use, rainfall and topography (Quinn 2004).

Scaling-up techniques have been noted as an important area of further research (Quinn 2002), although these are often associated with considerable uncertainty (Heathwaite 2003). Connectivity between land and waterways has been noted as important. Modelling connectivity has been achieved by models such as P and N index models (Heathwaite *et al.* 2000), where site nutrient loss risk is ranked by source and transport factors. Linking catchment nutrient export with water quality remains under-researched, particularly at edge-of-field (Heathwaite 2003), but studies in this area are increasing (Kurz *et al.* 2005). Similarly, in Australia there has been little research on N processes for off-site losses to stream loadings (Hunter 2000). One of the major limitations to understanding nutrient cycling at larger scales is the inability to scale up because there has been limited research at larger scales worldwide (Gillingham and Thorrold 2000; Hunter 2000).

Studies have only recently linked scales such as from small plot to paddock to large catchments (Haygarth *et al.* 2005), or from plot or paddock to farm scales (Cornish *et al.* 2002; Barlow *et al.* 2005). Although loads were not reported, concentrations of P from pasture runoff were dependent on scale, but the effect was small (Cornish *et al.* 2002). In contrast, Barlow *et al.* (2005) showed that under flood irrigation, paddock-scale P exports could not be used directly to estimate farm-scale exports because P concentrations and loads decreased between paddock and farm scales. The transport between paddocks, drainage channels (Barlow *et al.* 2003) and streams, and therefore riparian assimilation, requires further research for use within catchment models. Next, we

discuss catchment models and these related issues in further detail.

#### **Modelling approaches**

This section discusses the use of catchment models by end users, and the ability of models to represent aspects of importance to catchment managers. The section also describes the types of models used in the assessment of nutrient pollution in Australian catchments by catchment managers.

A prime consideration in catchment modelling is the end user of the model. Newham *et al.* (2004a), in the context of contaminant modelling, found end users sought models capable of simulating effects of land-use change, management of riparian zones, flow management and point-source control. End users also wished to apply models over a range of scales, use models to assist in communication and provide estimates of uncertainty. Priority was given to estimate sediment, TP, salt, TN and filterable P fluxes. Of note is that managers ranked riparian-zone management as the most important area for future research. Effort in this area is supported by Rutherford *et al.* (2003), who identified deficiencies in the simulation of riparian zone and in-stream nutrient processing in catchment models. Similar issues are discussed by Caminiti (2004).

Model applicability to new catchments and relevance to end users was also considered critical by Newham *et al.* (2005a). Suggestions for incorporation into future model development included use of a daily time step to enable coupling of qualitative models to assess ecosystem responses to contaminant loads, scenario generation and visualisation features to communicate outputs. To meet requirements of end-users such as catchment managers and government agencies, Heathwaite (2003) suggested that effort in pollutant export modelling needs to shift towards developing models that are simple to use and easy to apply. Catchment managers must also be made more aware of the inherent limitation in the use of specific models.

There are many categories of models (Jakeman *et al.* 2006) but in water-quality modelling there are three main categories: empirical, conceptual and physics-based. Some models may contain a mix of modules from these categories, so the distinction is not always sharp.

#### *Empirical models*

Empirical models are based on deriving responses from observations of data and are generally the simplest of the models. They are useful to identify sources of sediment and nutrient generation, but are catchment specific (Letcher *et al.* 1999; Merritt *et al.* 2003). If relationships with potential drivers can be derived, however, broader application of empirical models could be applied. Such models are generally not event-responsive and take no account of spatial distribution. Letcher *et al.* (1999) described that

many empirical models are based on statistical analysis of catchment data and are therefore more suitable for within-catchment analysis. Moreover, Donnelly *et al.* (1998) noted that, although empirical models can give accurate predictions under conditions representative of original datasets, little insight may be obtained into the causes of the observed relationships. Nutrient-generation rate or export-coefficient rate approaches are examples of empirical approaches. The USLE (Universal Soil Loss Equation) and CMSS are examples of widely used empirical models.

#### *Conceptual models*

Conceptual (process-based) models are typically represented by a linked configuration of internal storages, and usually include transfer mechanisms (Donnelly *et al.* 1998; Merritt *et al.* 2003). In contrast to empirical models, conceptual models reflect hypotheses about the processes in the system. These models can provide an indication of qualitative and quantitative effects of land-use change without the need for large amounts of spatially and temporally distributed input data. Conceptual models are typically calibrated against observed data including stream discharge and nutrient concentrations, but may have several 'best' parameter sets (Viney *et al.* 2000; Merritt *et al.* 2003). Conceptual models are a practical compromise between physics-based and empirical models, and are suited to long-term prediction in large catchments (Viney *et al.* 2000). Conceptual models do not require large datasets that are often required by physics-based models, which may not be appropriate at large catchment scale (Viney *et al.* 2000).

#### *Physics-based models*

Physics-based models are often derived at small plot scale under specific conditions, but in practice are also used at much larger scales, which has drawn criticism as to their applicability (Donnelly *et al.* 1998; Viney *et al.* 2000; Merritt *et al.* 2003). For example, the physics-based soil erosion and transport model, Watershed Erosion Prediction Project (WEPP), developed in the USA, requires detailed knowledge and a large number of inputs (Merritt *et al.* 2003). Therefore, model complexity, computational requirements and associated error accumulation of physics-based models can limit their applicability for estimating catchment exports (Letcher *et al.* 1999; Merritt *et al.* 2003). This is a particular problem in data-sparse catchments, common in Australia. Consequently, we shall base our discussion on non-physics-based models, such as empirical and conceptual models that are deemed more suitable for use by catchment managers.

#### *Discussion of model types*

The choice of model was suggested by Merritt *et al.* (2003) to be mainly governed by the emphasis of researchers and model developers. A common misconception among model users is

that model accuracy increases with complexity (Letcher *et al.* 1999; Merritt *et al.* 2003). Complex models may suffer from error accumulation and lack of readily available input data. In contrast, simple models may predict equally as well or better than complex models (Jakeman and Hornberger 1993), including estimating nutrient exports in catchments. Similarly, empirical and conceptual approaches can be combined into models that allow investigation of catchment source strengths, sensitivity to climate variability and event responsiveness, and have good general interpretability of results (Letcher *et al.* 1999; Merritt *et al.* 2003).

Predictions from water-quality models are subject to considerable absolute error, and should therefore be used with caution when determining absolute quantities, although their ability to rank alternative management strategies is much more reliable (Moore and Gallant 1991). Croke and Jakeman (2001) noted that further progress is needed on the difficult problem of assessing land-use effects within catchment models. In addition, current catchment models do not typically incorporate effects of land-use changes over time. The following section discusses selected catchment models.

#### **Catchment-scale models for evaluating nutrient export**

This section presents an overview of selected models used by catchment managers to estimate nutrient exports in Australian catchments. It is not our intention to include all models. Simple models have been favoured for broad assessment purposes over complex models requiring significant data inputs. It needs to be noted that models should be used with appropriate event-based water-quality data (Hunter 2000; Letcher *et al.* 2002), because routine water quality monitoring often does not capture large runoff events that may carry a large percentage of the load (Baginska *et al.* 2003). Modelling can therefore be a useful approach to assess diffuse source nutrient generation from catchments, or test management changes. However, nutrient loads expressed as annual averages are likely to underestimate the ecological impact on coastal lagoons from episodic events (Webster and Harris 2004). There is a need to identify appropriate nitrogen models to use in conjunction with catchment models, and to identify diffuse sources of nutrients and contributions of various processes to catchment-scale nutrient export, and the ecologically sustainable level of N use (Rose *et al.* 2000). In the next section, we discuss the CMSS model, CatchMODS (Catchment scale Management Of Diffuse Sources model) and overview the EMSS (Environmental Management Support System) and LASCAM (LARGE SCAle CATCHment Model) models.

#### *Catchment Management Support System (CMSS)*

The effects of land use and management policies on long-term nutrient loads delivered to rivers have been captured in the Catchment Management Support System (CMSS), designed

to assist catchment managers (Marston *et al.* 1995; Davis and Farley 1997). CMSS has been widely used in Australia as an initial planning tool because of its simplicity, ease of use and ease of results presentation (Gourley *et al.* 1996; Davis *et al.* 1998a).

The program structure includes four modules:

- a database module describing catchment land uses, spatial attributes, nutrient generation rates and management practices;
- a policy module that allows the user to set up and modify policy sets;
- a predictive module; and
- an interrogation module that allows the user to examine the basis of load and cost predictions, where data was obtained and how variable or relevant it is.

The predictive module calculates nutrient loads by summation of total area per land use within the catchment with nutrient generation rate per unit area. There is also allowance for point sources (Davis and Farley 1997; Letcher *et al.* 2002). An expert system is provided to help narrow the range of nutrient export rates, although only a range of values is generated, reflecting inherent variability and a lack of knowledge of export rates (Letcher *et al.* 1999). In addition, CMSS assumes each land-use generation rate is uniform. However, soil fertility and land-management practices often vary widely within land uses.

CMSS does not model the hydrology of the catchment (Letcher *et al.* 1999). This is one of the major limitations of this model, given the importance of flow when estimating nutrient exports (Rosich and Cullen 1982; Cosser 1989; Letcher *et al.* 2002) and understanding the release and transport of nutrients. However, more accurate representation can be gained by using measured loads, and then comparing loads derived from CMSS (Davis and Farley 1997; Joo *et al.* 2000). When the CMSS stream routing and assimilation functions are not used, CMSS can overestimate nutrient loads (Joo *et al.* 2000; Letcher *et al.* 2002), thus particular care is needed for application of the model to large areas (Baginska *et al.* 2003). It has also been noted that unit-area models such as CMSS are indicative of long-term nutrient generation, and therefore may not compare well with measured annual loads for a particular year due to high variability of rainfall and runoff (Baginska *et al.* 2003). This is particularly important when interpreting short-term nutrient generation studies. Letcher *et al.* (2002) pointed out that, in general, CMSS is not used to provide an accurate estimate of loads, but rather to provide preliminary information of relative source strengths of different land use and management options.

### CatchMODS

CatchMODS (Catchment scale Management Of Diffuse Sources model) is an integrated modelling framework

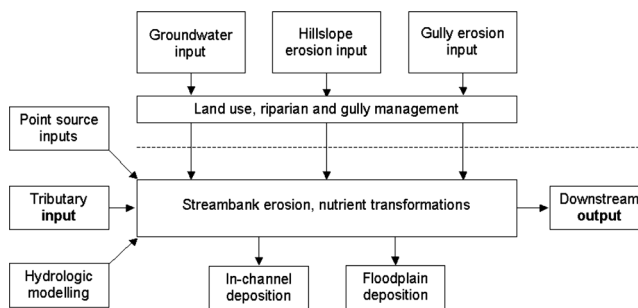


Fig. 3. CatchMODS framework showing components in a stream reach (Newham *et al.* 2004b).

designed to simulate and assess catchment-scale land management and thus reduce nutrient and sediment delivery (Newham *et al.* 2004b). The model structure integrates linked hydrological, sediment and nutrient export models (Fig. 3). Economic assessment of management options is also a feature, and is increasingly required by end users (Argent 2004). The modelling framework is designed to allow identification of priority subcatchments for management intervention, simulation of hydrological processes and climate variation, identification of critical source areas (parts of catchments contributing high loads to streams) and simulation of management changes (Newham *et al.* 2004b). Although riparian revegetation and gully management is modelled, management simulation of riparian buffer zones is not (Newham *et al.* 2005a).

The framework includes spatial representation based on lumped modelling at linked river reaches and subcatchment units. The sediment submodel is based on the SedNet model (Prosser *et al.* 2001). This component includes estimation of gully, hillslope and stream bank erosion sources. Several enhancements to the SedNet-based sediment submodel have been implemented to improve sediment flux predictions, including: (i) estimating erosion rates and gully dimensions based on severity classes; (ii) improving the quality of spatial data inputs; and (iii) quantifying sediment sources from contemporary rather than historical-based generation rates.

The hydrological submodel used is based on the conceptual IHACRES (Identification of unit Hydrographs And Component flows from Rainfall, Evapotranspiration and Stream flow data) rainfall–runoff model (Jakeman and Hornberger 1993). The quality of predictions using IHACRES, as with any such model, is influenced by rain gauge density, stream gauge rating quality and catchment response dynamics, particularly baseflow (Hansen *et al.* 1996). The more recent catchment–moisture deficit version of the model, with its low level of complexity (6 parameters), reliably reproduces measured hydrographs and is therefore useful in data-sparse catchments (Croke and Jakeman 2004).

The nutrient modelling component simulates TN by using three sources of N, viz. sediment-associated, groundwater-associated and point source inputs. Nitrogen losses in stream

reaches are estimated by an exponential decay using channel area, although this requires further testing (Newham *et al.* 2004b). Total phosphorus export is estimated directly from observed suspended sediment load, based on the assumption that P is transported adsorbed on sediment particles. However, total N and P losses from intensive farmland are likely to be underestimated given the current reliance on erosion submodels. Consequently, representation of dissolved nutrient export, as discussed in this review, should be taken into account.

### EMSS

The Environmental Management Support System (EMSS) is a software tool developed to aid water-quality management in south-east Queensland catchments (Vertessy *et al.* 2001).

The model structure comprises a lumped conceptual rainfall–runoff and pollutant export model, a flow and pollutant routing model and a reservoir storage module (Vertessy *et al.* 2001; Merritt *et al.* 2003). The reservoir model simulates regulation of river flows and traps pollutants. The model uses Geographic Information Systems (GIS) layers of sub-catchment boundaries and grids of land use, daily rainfall and evapotranspiration to predict daily loads of suspended sediment, TP, TN and pathogens for individual subcatchments and routes these through a river network (Vertessy *et al.* 2001).

In contrast to annual pollutant load reporting in CatchMODS, pollutant loads in EMSS are predicted daily by the hydrological model, but reported monthly to reduce apparent daily errors (Vertessy *et al.* 2001). Loads are predicted by the event mean concentration (EMC) and the baseflow runoff volume by dry weather pollutant concentration (DWC) (Cuddy *et al.* 2001; Merritt *et al.* 2003). Flow components are multiplied by these user-specified EMC and DWC factors to estimate daily loads (Newham *et al.* 2005a). Different EMC and DWC values can be allocated to subcatchments, depending on land use (Vertessy *et al.* 2001), in a similar fashion to CMSS. However, EMC and DWC have been noted to be highly variable, with further data needed to assist further development of the model.

### LASCAM

A conceptual model for N and P mobilisation and transport was developed by Viney *et al.* (2000). The hydrological model component, LASCAM, has 37 parameters for the water balance and 6 for sediment transport, although these can be reduced by using literature values (Viney and Sivapalan 2001). The model framework embraces conceptual nutrient balance models, including soluble and particulate forms, with 11 parameters for P and a further 18 for N. However, this component is less likely to suffer from problems associated with identifiability compared with more complex models (Merritt *et al.* 2003). The large-scale lumped nature of the model application suggests that complex representation was not needed (Viney *et al.* 2000; Merritt *et al.* 2003).

A disaggregation–aggregation approach, which involves disaggregating catchment-scale variables to point scale, links catchment-scale to small-scale physics (Viney and Sivapalan 2004). A disadvantage is that the model is based on the assumption that there is no significant nutrient uptake by riparian vegetation. Although at least 10 years data of frequently collected river nutrient concentrations were available to calibrate the model, nitrate was concluded to be poorly predicted (Viney and Sivapalan 2001).

### Further considerations for catchment nutrient export modelling

There are several further considerations when modelling catchment nutrient export to waterways. Even when simple models are used for estimation of nutrient loads, collection of basic data on climate, flow and pollutant concentration is recommended for at least a few years (Letcher *et al.* 2002). Ideally, monitoring programs should include nested stream gauges to allow in-stream processes to be understood, well distributed climate stations, and high temporal frequency monitoring to better understand storm event nutrient generation (Letcher *et al.* 2002). Letcher *et al.* (2002) also suggested using an alternative modelling technique, namely regionalisation, that uses subcatchment information to infer model parameters by calibration and to derive landscape attributes from spatial information.

The importance of moving research from detailed plot to catchment scale, or from site-specific models to generalisations applicable to catchments, was emphasised by Davis and Koop (2006). Worldwide, these concerns have also been expressed by McDowell *et al.* (2004), and although simulation approaches may be adequately represented at paddock or slope scale, processes representing P export at catchment scale are generally not well represented (Gillingham and Thorrold 2000). Processes such as stream bank and gully erosion, macropore subsurface flow and floodplain processes are also important in delivering nutrients to waterways. The importance of including these processes and nutrient species in user-friendly models was highlighted by Davis and Koop (2006). Further research is also required to simulate riparian zone delivery – a key requirement of end-users as discussed previously in this review. Concerns that riparian buffer strip effectiveness for removing P may decline over time (Gillingham and Thorrold 2000) should also be addressed. Finally, there is also a lack of knowledge on the economic effectiveness of best management practices from an environmental perspective (McDowell *et al.* 2004).

### Conclusions

This review has linked current knowledge of N and P generation with future needs for catchment models. It has

highlighted the importance of dissolved N and P for overland flow and groundwater pathways for sheep, beef and dairy grazing land use. Given the amounts of dissolved N and P transport in these pathways from some land uses, the effectiveness of riparian buffers to remove N and P may not be adequate. Rainfall and hydrology, and dissolved N and P losses from pastures, are therefore important factors that should be incorporated into catchment models. Although the Australian environment can be very different from other regions, our review shows it is likely that nutrient losses under Australian dairying conditions have many similarities to worldwide studies, although management and hydrological differences make direct comparisons difficult. There have also been few published studies in Australia examining environmental losses of N and P from dairy effluent application, with further research required. Similarly, uniform nutrient generation rates commonly used in some catchment models may not always be appropriate because management practices vary considerably within land uses.

Further research linking nutrient movement from paddocks to streams is warranted. We suggest that riparian processes that should be considered in catchment modelling include overland by-pass flow and rainfall intensity and soil macropore changes after animal exclusion. Similarly, one of the major limitations to understanding nutrient cycling at larger scales is the inability to scale-up, although research is now increasing in this area. Further research of downstream nutrient assimilation and subsequent water quality within large catchments is needed. Our review has highlighted the importance of nitrate leaching as a pathway under Australian grazing conditions. Research on this topic in Australian catchments is needed because diffuse nitrogen generation from agriculture has generally not been well quantified. It is therefore of some concern that catchment models used by catchment managers may not adequately simulate N and P export and riparian processes from pastoral land. Adequate riparian process simulation is a major model requirement of catchment managers. Catchment models need to represent catchment-event-based loads, intensively farmed land use, management and forms of nutrients. Otherwise there is a likelihood of either underestimating nutrient losses, or potentially overestimating the effectiveness of riparian buffers.

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