CSIRO PUBLISHING
Review
www.publish.csiro.au/journals/mfr
*Marine and Freshwater Research* 2006, **58**, 757–774

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

*J. J. Drewry*[A,B,C,E], *L. T. H. Newham*[A], *R. S. B. Greene*[B,C], *A. J. Jakeman*[A,B] and *B. F. W. Croke*[A,B,D]

[A]Integrated Catchment Assessment and Management Centre, The Australian National University, Canberra, ACT 0200, Australia.
[B]Cooperative Research Centre for Landscape Environments and Mineral Exploration, The Australian National University, Canberra, ACT 0200, Australia.
[C]School of Resources, Environment and Society, The Australian National University, Canberra, ACT 0200, Australia.
[D]Department of Mathematics, The Australian National University, Canberra, ACT 0200, Australia.
[E]Corresponding author. Email: john.drewry@anu.edu.au

*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*. 1998*b*; 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).


10.1071/MF05166 1323-1650/06/080757

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*. 2004*b*).

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-incidental) and increased soil fertility via soil additions of fertiliser, manure or dung (incidental). 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-$\mu$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 $ha^{-1}$ 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 $ha^{-1}$ $year^{-1}$ 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 $ha^{-1}$ $year^{-1}$) | |
|---|---|---|---|
| | | TP | TN |
| Mixed farming or rural | 5 | 0.2, 0.3, 0.57 | 4.5 |
| Forest/scrub | 6 | 0.03, 0.05[A], 0.07[A] | 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[B] | 3.2, 3.7, 5.3, 22.8[B] |

[A]Plantation forest (*Pinus* species).
[B]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 kg N $ha^{-1}$ $year^{-1}$ were recorded in grass/clover pastures with no incidental N application (Tyson *et al.* 1997). However, mean leaching losses increased to 50 kg N $ha^{-1}$ $year^{-1}$ in pastures fertilised with 200 kg N $ha^{-1}$ $year^{-1}$ and ranging from 29 to 72 kg N $ha^{-1}$ $year^{-1}$ (Tyson *et al.* 1997). Although leaching losses at the same UK site increased greatly with 400 kg N $ha^{-1}$ $year^{-1}$ 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 (0.01–0.56 kg P $ha^{-1}$ $year^{-1}$) 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 (kg P $ha^{-1}$ $year^{-1}$) | Dissolved P (kg P $ha^{-1}$ $year^{-1}$) | TP (mg $L^{-1}$) | Dissolved P (mg $L^{-1}$) | TN (kg N $ha^{-1}$ $year^{-1}$) | $NO_3$ (kg $NO_3$-N $ha^{-1}$ $year^{-1}$) | TN (mg $L^{-1}$) | $NO_3$ (mg $L^{-1}$) | Size | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Leaching | Vic, Australia | | | | | | 0–19 | | 0–24 | Plots | Ridley *et al.* (2001) |
| Leaching | Vic, Australia | 0.09–0.45 | | | | | 0.3–10 | | 3–26 | 3–17 ha | Ridley *et al.* (2003) |
| Leaching | England | | | | | | 6–34 (0 N fert) | | | 0.4 ha plots | Cuttle *et al.* (1998) |
| Leaching | England | | | | | | 2–46 (N fert) | | | 0.4 ha plots | Cuttle *et al.* (1998) |
| Leaching | England | | | | | | 12.9 (0 N fert) | | | | Tyson *et al.* (1997) |
| Leaching | England | | | | | | 50 (200 kg $ha^{-1}$ N fert) | | | | Tyson *et al.* (1997) |
| Drainage through flow | SA, Australia | To 0.14 | 70% | 0.1–0.5 | | | To 0.8 | | To 19 | Plots | Cox and Pitman (2001) |
| OF & through flow | SA, Australia | To ~0.20 | 8–50% | 0.13–1.02 | 0.01–0.53 | | ~0.15–0.25 | | 0.6–1.79 | 2–4 ha | Stevens *et al.* (1999) |
| OF | Australia | 0.01–0.56 | | Various | | | | | Various | Various sites | McCaskill *et al.* (2003) |
| OF | Vic, Australia | | | | | | 0–2 | | 0.3–16.1 | Plots | Ridley *et al.* (2001) |
| OF | Vic, Australia | 0.04–0.47 | | 0.12–0.92 | | | 0.7–3.8 | | 3–8.2 | 3–17 ha | Ridley *et al.* (2003) |
| OF | SA, Australia | To 0.07 | 35% | 0.1–0.5 | | | 0–0.5 | | 0–0.3 | Plots | Cox and Pitman (2001) |
| OF | Ireland | | 0.69–4.7 (77%) | | 0.09–1.0 | | | | | Plots | Kurz *et al.* (2005) |
| OF and interflow | England | 3 | 32% | 0.12 | 0.04 | | | | | 1 ha plots | Haygarth and Jarvis (1997) |
| Stream | SA, Australia | 1.04 | 0.14 | 0.35 | 0.048 | 9.0 | 4.1 | 1.68 | 1.38 | 3 $km^2$ | Nelson *et al.* (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) |

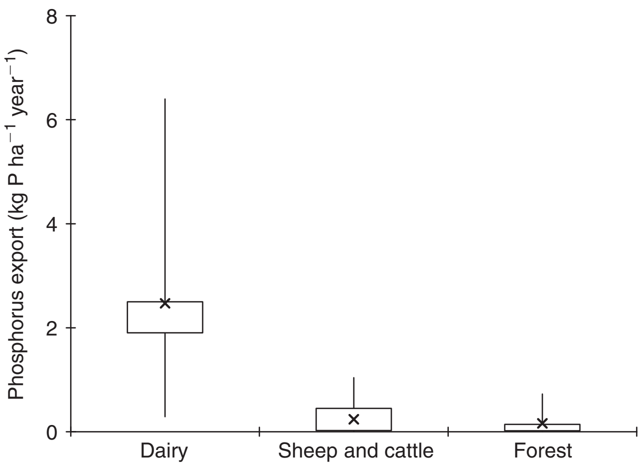


**Fig. 1.** Phosphorus export rates (kg P ha$^{-1}$ year$^{-1}$) for Australian field studies. The box is between lower and upper quartile, whiskers are maximum and minimum, × is the mean.

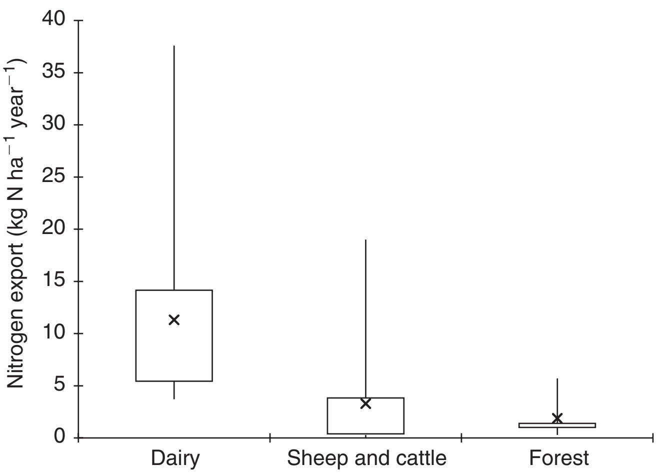


**Fig. 2.** Nitrogen export rates (kg N ha$^{-1}$ year$^{-1}$) for Australian field studies. The box is between lower and upper quartile, whiskers are maximum and minimum, × is the mean.

## 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 (kg P ha$^{-1}$ year$^{-1}$) | Dissolved P (kg P ha$^{-1}$ year$^{-1}$) | TP (mg L$^{-1}$) | Dissolved P (mg L$^{-1}$) | Annual P fertiliser (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 *et al.* (2001) |
| OF | SA, Australia | 0.29 | 0.14 (48%) | | | 15 | | 2.6 ha | Fleming *et al.* (2001) |
| OF | NSW, Australia | 1.9 | 56% | 0.95 | | 0 | | 4 ha | Cornish *et al.* (2002) |
| OF | NSW, Australia | 2.5 | 59% | 0.66 | | 0 | | 140 ha | Cornish *et al.* (2002) |
| OF on drained plots | New Zealand | 0.031 | 0.009 | | | 55–75[A] | 0–400 kg N ha$^{-1}$ | 0.09 ha | Smith and Monaghan (2003) |
| OF on undrained plots | New Zealand | 0.23 | 0.09 | | | 55–75[A] | 0–400 kg N ha$^{-1}$ | 0.05 ha | Smith and Monaghan (2003) |

[A]75 kg year 1; 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 $ha^{-1}$ $year^{-1}$) | Dissolved P (kg P $ha^{-1}$ $year^{-1}$) | TP (mg $L^{-1}$) | Dissolved P (mg $L^{-1}$) | Cows $ha^{-1}$ | Soil Olsen P (mg $kg^{-1}$) | Annual P fertiliser (kg P $ha^{-1}$) | Comment | Size | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| Vic, Australia | 3.6–23 | 80–95% | 3–11 | 3–9 | 3.4 | | 0, 20, or 29 | Flood irrigation re-use | Paddock | Barlow *et al.* (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 *et al.* (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 *et al.* (2003) |
| Vic, Australia | 0.84–1.08 | | | | | 33 | 45 | 4 flood irrigations, grazed once | Paddock | Mundy *et al.* (2003) |
| Vic, Australia | 6–17[A] | | 1.7–2.8 (3.3–20[A]) | 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 *et al.* (2004) |

[A]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 $ha^{-1}$ $year^{-1}$) | $NO_3$ (kg $NO_3$-N $ha^{-1}$ $year^{-1}$) | TN (mg $L^{-1}$) | Comment | Size | Reference |
|---|---|---|---|---|---|---|---|
| OF | New Zealand | 0.046[A] | 0.023 | | Drained plots 0 kg N $ha^{-1}$ | 0.09 ha | Smith and Monaghan (2003) |
| OF | New Zealand | 0.084[A] | 0.036 | | Drained plots 400 kg N $ha^{-1}$ | 0.09 ha | Smith and Monaghan (2003) |
| OF | New Zealand | 0.333[A] | 0.19 | | Un-drained plots 0 kg N $ha^{-1}$ | 0.05 ha | Smith and Monaghan (2003) |
| OF | New Zealand | 0.841[A] | 0.48 | | Un-drained plots 400 kg N $ha^{-1}$ | 0.05 ha | Smith and Monaghan (2003) |
| OF | New Zealand | 2.0 | | | 7 Flood irrigation events, each grazed | 0.8 ha | Carey *et al.* (2004) |
| OF | Vic, Australia | 0.23–0.4[B] | | 1.14–1.91 | 1 Flood irrigation only, un-grazed | | Mundy *et al.* (2003) |
| OF | Vic, Australia | 1.31–2.35[C] | | | 4 Flood irrigations only, grazed once | | Mundy *et al.* (2003) |
| OF | Vic, Australia | 10–16 | | 2.8–6.2 (3.1–17[D]) | 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 $km^2$ | Nelson *et al.* (1996) |
| Stream | NSW, Australia | 4.1, 4 | | | Dryland dairy | 4–200 ha | Hollinger and Cornish (2002) and Baginska *et al.* (1998) |
| Stream | NSW, Australia | 5.8, 6 | | | Irrigated dairy | 44 ha | Hollinger and Cornish (2002) and Baginska *et al.* (1998) |
| Stream | New Zealand | 35.3 | 30.7 | | Dairy catchment | 15 $km^2$ | Wilcock *et al.* (1999) |

[A]Nitrate + ammonium only.
[B]1 flood irrigation only.
[C]Total inorganic N.
[D]Following fertiliser application.

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

| Region | $NO_3$ (kg $NO_3$-N ha$^{-1}$ year$^{-1}$) | $NO_3$ (mg L$^{-1}$) | N fertiliser application (kg N ha$^{-1}$ year$^{-1}$) | Size | Reference |
|---|---|---|---|---|---|
| Vic, Australia | 3.7–14.6 | 1.9–4.7 | 0 | 0.062 ha | Eckard *et al*. (2004) |
| Vic, Australia | 6.2–22 | 3–5.6 | 200 urea | 0.062 ha | Eckard *et al*. (2004) |
| Vic, Australia | 4.3–37.6 | 2.3–11.4 | 200 ammonium nitrate | 0.062 ha | Eckard *et al*. (2004) |
| New Zealand | 40 | 7 | 0 | Farmlet | Ledgard *et al*. (1998) |
| New Zealand | 81 | 10 | 200 | Farmlet | Ledgard *et al*. (1998) |
| New Zealand | 25 (TN 26.5) | 6.9 (TN 7.3) | 0 | 0.09 ha | Monaghan *et al*. (2002) |
| New Zealand | 30 | 7.9 | 0 | 0.09 ha | Monaghan *et al*. (2005) |
| New Zealand | 46 | 12.3 | 200 | 0.09 ha | Monaghan *et al*. (2005) |
| New Zealand | 56 | 15 | 400 | 0.09 ha | Monaghan *et al*. (2005) |

(up to 2.3 kg P ha$^{-1}$) occurring during the wettest year, whereas losses in dry years were up to ~0.5 kg P ha$^{-1}$ (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 $NO_3$-N ha$^{-1}$ year$^{-1}$ 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$^{-1}$ year$^{-1}$ under 200 and 400 kg urea N applied ha$^{-1}$ year$^{-1}$, respectively (Ledgard *et al*. 1998). Similarly, nitrate losses of 46 and 56 kg N ha$^{-1}$ year$^{-1}$ were leached under 200 and 400 kg urea N applied ha$^{-1}$ year$^{-1}$, 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 $ha^{-1}$ $year^{-1}$) | Dissolved P (kg P $ha^{-1}$ $year^{-1}$) | TP (mg $L^{-1}$) | Comment | Size | Reference |
|---|---|---|---|---|---|---|---|
| Leaching | New Zealand | 0.16–0.43 | 0.05–0.11 | 0.1–0.2 | 0–400 kg N $ha^{-1}$ | 0.09 ha | Monaghan *et al.* (2005) |
| Leaching | New Zealand | 0.15 | 0.059 (39%) | | 0 kg N $ha^{-1}$ | 0.09 ha | Monaghan *et al.* (2002) |
| Stream | SA, Australia | 1.11 | 0.84 | 0.4 | Sand over clay | 3 $km^2$ | Nelson *et al.* (1996) |
| Stream | NSW, Australia | 1.9–2.5 | | 1.5 EMC (to 5) | Dryland dairy | 4 ha | Hollinger and Cornish (2002) and Baginska *et al.* (1998) |
| Stream | NSW, Australia | 6–6.4 | | 4 EMC | Irrigated dairy | 44 ha | Hollinger and Cornish (2002) and Baginska *et al.* (1998) |
| Stream | New Zealand | 1.16 | 0.54 | | Dairy catchment | 15 $km^2$ | Wilcock *et al.* (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 $ha^{-1}$ and 1 kg total nitrogen (TN) $ha^{-1}$ 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 $ha^{-1}$ and 2.3–2.8 kg TN $ha^{-1}$ (Chessman 1986). Nutrient loss may also be less in plantation than in native forest (Cooper and Thomsen 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 $ha^{-1}$ $year^{-1}$) | TN (kg N $ha^{-1}$ $year^{-1}$) | $NO_3$ (kg $NO_3$-N $ha^{-1}$ $year^{-1}$) | TN (mg $L^{-1}$) | Comment | Size | Reference |
|---|---|---|---|---|---|---|---|
| NSW, Australia | (0.01–0.07) | (0.2–1.1) | | 0.25–0.26$^{A}$ | Bushland | 25–174 $km^2$ | Hollinger and Cornish (2002) |
| Qld, Australia | | | | 0.24$^{B}$ | Forest | | Bramley and Roth (2002) |
| NSW, Australia | (0.2–0.73) | (1.4–5.7) | | | 7 subtropical catchments, 50–94% forest | 43–17 000 $km^2$ | Eyre and Pont (2003) |
| Vic, Australia | 0.024 | | | | All forest, some regrowth. Eucalyptus, Nothofagus | 13 $km^2$ | Campbell (1978) |
| Vic, Australia | 0.083 | | | | 65% forest (Eucalyptus), some logging, 35% pasture | 140 $km^2$ | Campbell (1978) |
| Vic, Australia | 0.01$^{C}$ | 1.0$^{C}$ | | | Wet sclerophyll forest (7% affected by fire) | 720 $km^2$ | Chessman (1986) |
| New Zealand | 0.09 | 1.31 | 0.55 | | Pine plantation | 0.34 $km^2$ | Cooper and Thomsen (1988) |
| New Zealand | 0.12 | 3.7 | 2.84 | | Native podocarp & hardwood | 0.28 $km^2$ | Cooper and Thomsen (1988) |
| Americas | | 5.1 (0.7–8) | 2.43 | | Tropical forest | Varied | Lewis *et al.* (1999) |
| USA | | | ~1–30 | | Forest (oak, maple) | 39–1156 $km^2$ | Swistock *et al.* (1997) |

$^{A}$Median monthly flow weighted concentration. $^{B}$Median values. $^{C}$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.* 2005*b*). 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*. (2004*a*), 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*. (2005*a*). 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.* 1998*a*).

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 routeing 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 designed to simulate and assess catchment-scale land management and thus reduce nutrient and sediment delivery (Newham *et al.* 2004*b*). 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.* 2004*b*). Although riparian revegetation and gully management is modelled, management simulation of riparian buffer zones is not (Newham *et al.* 2005*a*).

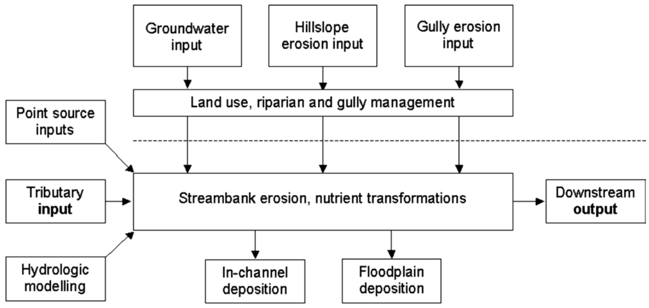


**Fig. 3.** CatchMODS framework showing components in a stream reach (Newham *et al.* 2004*b*).

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.* 2004*b*). 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 routeing 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 subcatchment 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.* 2005*a*). 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.

## Acknowledgments

This review was undertaken with funding from The Australian National University and the Cooperative Research Centre for Landscape Environments and Mineral Exploration. Thanks to the reviewers for helpful comments on the manuscript.

## References

Allingham, K. D., Cartwright, R., Donaghy, D., Conway, J. S., Goulding, K. W. T., and Jarvis, S. C. (2002). Nitrate leaching losses and their control in a mixed farm system in the Cotswold Hills, England. *Soil Use and Management* **18**, 421–427. doi:10.1079/SUM2002159

Anon (1998). Macalister irrigation district nutrient reduction plan. Department of Natural Resources and Environment, Maffra, Australia.

ANZECC (2000). Australian and New Zealand guidelines for fresh and marine water quality. Volumes 1 and 2. Australian and New Zealand Environment and Conservation Council, Canberra.

Argent, R. M. (2004). An overview of model integration for environmental applications – components, frameworks and semantics. *Environmental Modelling & Software* **19**, 219–234. doi:10.1016/S1364-8152(03)00150-6

Baginska, B., Cornish, P. S., Hollinger, E., Kuczera, G., and Jones, D. (1998). Nutrient export from rural land in the Hawkesbury–Nepean catchment. In 'Proceedings of the 9th Australian Agronomy Conference, Wagga Wagga, 20–23 July 1998'. (Eds D. L. Michalk and J. E. Pratley.) pp. 753–756. (The Australian Society of Agronomy: Gosford, Australia.)

Baginska, B., Pritchard, T., and Krogh, M. (2003). Roles of land use resolution and unit-area load rates in assessment of diffuse nutrient emissions. *Journal of Environmental Management* **69**, 39–46. doi:10.1016/S0301-4797(03)00112-9

Barlow, K., Nash, D., Turral, H., and Grayson, R. (2003). Phosphorus uptake and release in surface drains. *Agricultural Water Management* **63**, 109–123. doi:10.1016/S0378-3774(03)00177-X

Barlow, K., Nash, D., and Grayson, R. B. (2005). Phosphorus export at the paddock, farm-section, and whole farm scale on an irrigated dairy farm in south-eastern Australia. *Australian Journal of Agricultural Research* **56**, 1–9. doi:10.1071/AR04166

Bramley, R. G. V., and Roth, C. H. (2002). Land-use effects on water quality in an intensively managed catchment in the Australian humid tropics. *Marine and Freshwater Research* **53**, 931–940. doi:10.1071/MF01242

Brodie, J., and Mitchell, A. (2005). Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. *Marine and Freshwater Research* **56**, 279–302. doi:10.1071/MF04081

Byun, D., Cho, Y., Huh, I., and Hart, D. E. (2005). Runoff-induced vertical thermal dynamics in a canyon-shaped reservoir during the summer monsoon. *Marine and Freshwater Research* **56**, 959–968. doi:10.1071/MF04285

Caitcheon, G. G., Olley, J. M., Hancock, G. J., and Wallbrink, P. J. (1999). Exposing the phosphorus myth: looking for evidence of anthropogenic phosphorus in large inland basins in Australia. In 'Proceedings of the International Conference on Diffuse Pollution, Perth, 16–20 May 1999'. (Eds C. Barber, R. Humphries and J. Dixon.) pp. 287–295. (CSIRO Land and Water: Canberra.)

Cameron, K. C., Di, H. J., and Condron, L. M. (2002). Nutrient and pesticide transfer from agricultural soils to water in New Zealand. In 'Agriculture, Hydrology and Water Quality'. (Eds P. M. Haygarth and S. C. Jarvis.) pp. 373–393. (CABI Publishing: Wallingford.)

Caminiti, J. E. (2004). Catchment modelling – a resource manager's perspective. *Environmental Modelling & Software* **19**, 991–997. doi:10.1016/J.ENVSOFT.2003.11.002

Campbell, I. C. (1978). Inputs and outputs of water and phosphorus from four Victorian catchments. *Australian Journal of Marine and Freshwater Research* **29**, 577–584. doi:10.1071/MF9780577

Carey, P. L., Drewry, J. J., Muirhead, R. W., and Monaghan, R. M. (2004). Potential for nutrient and faecal bacteria losses from a dairy pasture under border-dyke irrigation: a case study. In 'Proceedings of the New Zealand Grassland Association, Vol. 66'. pp. 141–149. (New Zealand Grassland Association Inc.: Dunedin.)

Chessman, B. C. (1986). Impact of the 1983 wildfires on river water quality in East Gippsland, Victoria. *Australian Journal of Marine and Freshwater Research* **37**, 399–420. doi:10.1071/MF9860399

Cooper, A. B., and Thomsen, C. E. (1988). Nitrogen and phosphorus in streamwaters from adjacent pasture, pine, and native forest catchments. *New Zealand Journal of Marine and Freshwater Research* **22**, 279–291.

Cooper, A. B., Smith, C. M., and Smith, M. J. (1995). Effects of riparian set-aside on soil characteristics in an agricultural landscape: implications for nutrient transport and retention. *Agriculture Ecosystems & Environment* **55**, 61–67. doi:10.1016/0167-8809(95)00605-R

Cornish, P. S., Hallissey, R., and Hollinger, E. (2002). Is a rainfall simulator useful for estimating phosphorus runoff from pastures – a question of scale-dependency? *Australian Journal of Experimental Agriculture* **42**, 953–959. doi:10.1071/EA01143

Cosser, P. R. (1989). Nutrient concentration-flow relationships and loads in the South Pine River, south-eastern Queensland. I. Phosphorus loads. *Australian Journal of Marine and Freshwater Research* **40**, 613–630. doi:10.1071/MF9890613

Cox, J. W., and Ashley, R. (2000). Water quality of gully drainage from texture-contrast soils in the Adelaide Hills in low rainfall years. *Australian Journal of Soil Research* **38**, 959–972. doi:10.1071/SR99106

Cox, J. W., and Pitman, A. (2001). Chemical concentrations of overland flow and throughflow from pastures on sloping texture-contrast soils. *Australian Journal of Agricultural Research* **52**, 211–220. doi:10.1071/AR00054

Cox, J. W., Kirkby, C. A., Chittleborough, D. J., Smythe, L. J., and Fleming, N. K. (2000). Mobility of phosphorus through intact soil cores collected from the Adelaide Hills, South Australia. *Australian Journal of Soil Research* **38**, 973–990. doi:10.1071/SR99125

Croke, B. F., and Jakeman, A. J. (2001). Predictions in catchment hydrology: an Australian perspective. *Marine and Freshwater Research* **52**, 65–79. doi:10.1071/MF00045

Croke, B. F., and Jakeman, A. J. (2004). A catchment moisture deficit module for the IHACRES rainfall-runoff model. *Environmental Modelling & Software* **19**, 1–5. doi:10.1016/J.ENVSOFT.2003.09.001

Croke, J., Hairsine, P., and Fogarty, P. (1999). Runoff generation and re-distribution in logged eucalyptus forests, south-eastern Australia. *Journal of Hydrology* **216**, 56–77. doi:10.1016/S0022-1694(98)00288-1

Cuddy, S. M., Rahman, J. M., Marston, F. M., Seaton, S. P., and Vertessy, R. A. (2001). Alternative approaches to incorporating management options into water quality modelling. In 'Proceedings of the Third Stream Management Conference: The Value of Healthy Streams, Brisbane, 27–29 August 2001'. (Eds I. Rutherfurd, F. Sheldon, G. Brierley and C. Kenyon.) pp. 169–174. (Cooperative Research Centre for Catchment Hydrology: Canberra.)

Cuttle, S. P., Scurlock, R. V., and Davies, B. M. S. (1998). A 6-year comparison of nitrate leaching from grass/clover and N-fertilized grass pastures grazed by sheep. *The Journal of Agricultural Science* **131**, 39–50. doi:10.1017/S0021859698005565

Davis, J. R., and Farley, T. F. N. (1997). CMSS: policy analysis software for catchment managers. *Environmental Modelling & Software* **12**, 197–210. doi:10.1016/S1364-8152(97)00012-1

Davis, J. R., and Koop, K. (2001). Current understanding of the eutrophication process in Australia. In 'Regional Management of Water Resources. Proceedings of a Symposium held during the Sixth IAHS Scientific Assembly, Maastricht, Netherlands, 18–27 July 2001'. (Eds A. H. Schumann, M. C. Acreman, R. Davis, M. A. Marino, D. Rosbjerg and X. Jun.) pp. 89–95. (IAHS Press: Wallingford.)

Davis, R., and Koop, K. (2006). Eutrophication in Australian rivers, reservoirs and estuaries – a southern hemisphere perspective on the science and its implications. *Hydrobiologia* **559**, 23–76. doi:10.1007/S10750-005-4429-2

Davis, J. R., Farley, T. F. N., Young, W. J., and Cuddy, S. M. (1998*a*). The experience of using a decision support system for nutrient management in Australia. *Water Science and Technology* **37**, 209–216. doi:10.1016/S0273-1223(98)00072-9

Davis, R., Hamblin, A., O'Loughlin, E., Austin, N., Banens, R., *et al.* (1998*b*). Phosphorus in the landscape: diffuse sources to surface waters. Occasional paper no 16/98. Land and Water Resources Research and Development Corporation, Canberra.

Di, H. J., and Cameron, K. C. (2000). Calculating nitrogen leaching losses and critical nitrogen application rates in dairy pasture systems using a semi-empirical model. *New Zealand Journal of Agricultural Research* **43**, 139–147.

Di, H. J., Cameron, K. C., Moore, S., and Smith, N. P. (1998). Nitrate leaching and pasture yields following the application of dairy shed effluent or ammonium fertiliser under spray or flood irrigation: results of a lysimeter study. *Soil Use and Management* **14**, 209–214.

Donnelly, T. H., Barnes, C. J., Wasson, R. J., Murray, A. S., and Short, D. L. (1998). Catchment phosphorus sources and algal blooms – an interpretative review. Technical Report 18/98. CSIRO Land and Water, Canberra.

Drewry, J. J. (2006). Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: a review. *Agriculture Ecosystems & Environment* **114**, 159–169. doi:10.1016/J.AGEE.2005.11.028

Drewry, J. J., McDowell, R. W., Monaghan, R. M., and Thorrold, B. S. (2003). Soil quality indicators in four dairy catchments: environmental and management implications. In 'Proceedings of the 22nd New Zealand Geographical Society Conference, Auckland, 6–11 July 2003'. (Eds J. Gao, R. Le Heron and J. Logie.) pp. 284–288. (The University of Auckland: Auckland.)

Eckard, R. J., White, R. E., Edis, R., Smith, A., and Chapman, D. F. (2004). Nitrate leaching from temperate perennial pastures grazed by dairy cows in south-eastern Australia. *Australian Journal of Agricultural Research* **55**, 911–920. doi:10.1071/AR04042

Eyre, B. D., and Pont, D. (2003). Intra- and inter-annual variability in the different forms of diffuse nitrogen and phosphorus delivered to seven sub-tropical east Australian estuaries. *Estuarine, Coastal and Shelf Science* **57**, 137–148. doi:10.1016/S0272-7714(02)00337-2

Fleming, N. K., and Cox, J. W. (1998). Chemical losses off dairy catchments located on a texture-contrast soil: carbon, phosphorus, sulfur, and other chemicals. *Australian Journal of Soil Research* **36**, 979–995. doi:10.1071/S98013

Fleming, N. K., and Cox, J. W. (2001). Carbon and phosphorus losses from dairy pasture in South Australia. *Australian Journal of Soil Research* **39**, 969–978. doi:10.1071/SR00042

Fleming, N. K., Cox, J. W., Chittleborough, D. J., and Dyson, P. (2001). An analysis of chemical loads and forms in overland flow from dairy pastures from in South Australia. *Hydrological Processes* **15**, 393–405. doi:10.1002/HYP.154

Ford, P. W., and Bormans, M. (2000). Nitrogen transformations in freshwater sediments. In 'Nitrogen Workshop 2000: Sources, Transformations, Effects and Management on Nitrogen in Freshwater Ecosystems, Melbourne'. (Eds B. T. Hart and M. R. Grace.) pp. 91–95. Occasional Paper 10/00. Land and Water Australia, Canberra.

Forsyth, A. R., Bubb, K. A., and Cox, M. E. (2006). Runoff, sediment loss and water quality from forest roads in a southeast Queensland coastal plain Pinus plantation. *Forest Ecology and Management* **221**, 194–206.

Gabric, A. J., and Bell, P. R. F. (1993). Review of the effects of non-point nutrient loading on coastal ecosystems. *Australian Journal of Marine and Freshwater Research* **44**, 261–283. doi:10.1071/MF9930261

Gillingham, A. G., and Thorrold, B. S. (2000). Review of New Zealand research measuring phosphorus in runoff from pasture. *Journal of Environmental Quality* **29**, 88–96.

Gourley, J., Cogle, A. L., Brebber, L., Herbert, B., Best, E., and Wright, N. (1996). Water quality and land use impacts in Lake Tinaroo/Barron River: IV. Decision support systems for use in the Tinaroo Dam catchment. In 'Downstream Effects of Land Use'. (Eds H. M. Hunter, A. G. Eyles and G. E. Rayment.) pp. 261–264. (Department of Natural Resources: Brisbane.)

Greene, R. S. B., and Hairsine, P. B. (2004). Elementary processes of soil-water interaction and thresholds in soil surface dynamics: a review. *Earth Surface Processes and Landforms* **29**, 1077–1091. doi:10.1002/ESP.1103

Hansen, D. P., Ye, W., Jakeman, A. J., Cooke, R., and Sharma, P. (1996). Analysis of the effect of rainfall and streamflow data quality and catchment dynamics on streamflow prediction using the rainfall-runoff model IHACRES. *Environmental Software* **11**, 193–202. doi:10.1016/S0266-9838(96)00048-2

Harris, G. P. (2001). Biochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns. *Marine and Freshwater Research* **52**, 139–149. doi:10.1071/MF00031

Hart, B. T., and Grace, M. R. (2000). Workshop report. In 'Nitrogen Workshop 2000: Sources, Transformations, Effects and Management on Nitrogen in Freshwater Ecosystems, Melbourne'. (Eds B. T. Hart and M. R. Grace.) pp. 5–29. Occasional Paper 10/00. Land and Water Australia, Canberra.

Haygarth, P. M., and Jarvis, S. C. (1997). Soil derived phosphorus in surface runoff from grazed grassland lysimeters. *Water Research* **31**, 140–148. doi:10.1016/S0043-1354(99)80002-5

Haygarth, P. M., and Jarvis, S. C. (1999). Transfer of phosphorus from agricultural soils. *Advances in Agronomy* **66**, 195–249.

Haygarth, P. M., and Sharpley, A. N. (2000). Terminology for phosphorus transfer. *Journal of Environmental Quality* **29**, 10–15.

Haygarth, P. M., Hepworth, L., and Jarvis, S. C. (1998). Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. *European Journal of Soil Science* **49**, 65–72. doi:10.1046/J.1365-2389.1998.00131.X

Haygarth, P. M., Wood, F. L., Heathwaite, A. L., and Butler, P. J. (2005). Phosphorus dynamics observed through increasing scales in a nested headwater-to-river channel study. *The Science of the Total Environment* **344**, 83–106.

Heathwaite, A. L. (2003). Making process-based knowledge useable at the operational level: a framework for modelling diffuse pollution from agricultural land. *Environmental Modelling & Software* **18**, 753–760. doi:10.1016/S1364-8152(03)00077-X

Heathwaite, L., Sharpley, A., and Gburek, W. (2000). A conceptual approach for integrating phosphorus and nitrogen management at watershed scales. *Journal of Environmental Quality* **29**, 158–166.

Holford, I. C. R., Hird, C., and Lawrie, R. (1997). Effects of animal effluents on the phosphorus sorption characteristics of soils. *Australian Journal of Soil Research* **35**, 365–373. doi:10.1071/S96048

Hollinger, E., and Cornish, P. (2002). Magnitude and management of diffuse agricultural nutrient pollution. In 'Coast to Coast, Tweed Heads, 4–8 November 2002'. pp. 190–193. (Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management: Brisbane.)

Houlbrooke, D. J., Horne, D. J., Hedley, M. J., Hanly, D. J., and Snow, V. O. (2004). A review of literature on the land treatment of farm-dairy effluent in New Zealand and its impact on water quality. *New Zealand Journal of Agricultural Research* **47**, 499–511.

Hunter, H. M. (2000). Agricultural sources of nitrogen to aquatic systems. In 'Nitrogen Workshop 2000: Sources, Transformations, Effects and Management on Nitrogen in Freshwater Ecosystems, Melbourne'. (Eds B. T. Hart and M. R. Grace.) pp. 74–79. Occasional Paper 10/00. Land and Water Australia, Canberra.

Hunter, H. M., and Armour, J. D. (2001). Offsite movement of nutrients: contrasting issues at three Australian study sites. In 'Proceedings of a Workshop on Offsite Movement of Agrochemicals in Tropical Sugar Production, Bundaberg, 8–9 May 2001'. pp. 68–76. (Department of Natural Resources and Mines: Brisbane.)

Jakeman, A. J., and Hornberger, G. M. (1993). How much complexity is warranted in a rainfall-runoff model? *Water Resources Research* **29**, 2637–2649. doi:10.1029/93WR00877

Jakeman, A. J., Letcher, R. A., and Norton, J. P. (2006). Ten iterative steps in development and evaluation of environmental models. *Environmental Modelling & Software* **21**, 602–614. doi:10.1016/J.ENVSOFT.2006.01.004

Johnston, W. H., Garden, D. L., Rancic, A., Koen, T. B., Dassanayake, K. B., *et al.* (2003). The impact of pasture development and grazing on water-yielding catchments in the Murray–Darling Basin in south-eastern Australia. *Australian Journal of Experimental Agriculture* **43**, 817–841. doi:10.1071/EA02236

Joo, M., McNeil, V. H., O'Brien, E., and Fearon, R. (2000). Nutrient emission modelling in the Dawson River Basin using Catchment Management Support System (CMSS). Department of Natural Resources, Brisbane.

Kirkby, C. A., Smythe, L. J., Cox, J. W., and Chittleborough, D. J. (1997). Phosphorus movement down a toposequence from a landscape with texture contrast soils. *Australian Journal of Soil Research* **35**, 399–417. doi:10.1071/S96045

Kuhnel, I., McMahon, T. A., Finlayson, B. L., Haines, A., Whetton, P. H., and Gibson, T. T. (1990). Climatic influences on streamflow variability: a comparison between southeastern Australia and southeastern United States of America. *Water Resources Research* **26**, 2483–2496. doi:10.1029/90WR00443

Kurz, I., Coxon, C., Tunney, H., and Ryan, D. (2005). Effects of grassland management practices and environmental conditions on nutrient concentrations in overland flow. *Journal of Hydrology* **304**, 35–50. doi:10.1016/J.JHYDROL.2004.07.022

Ledgard, S. F., Crush, J. R., and Penno, J. W. (1998). Environmental impacts of different nitrogen inputs on dairy farms and implications for the Resource Management Act of New Zealand. *Environmental Pollution* **102**, 515–519. doi:10.1016/S0269-7491(98)80077-8

Ledgard, S. F., Penno, J. W., and Sprosen, M. S. (1999). Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. *The Journal of Agricultural Science* **132**, 215–225. doi:10.1017/S002185969800625X

Letcher, R. A., Jakeman, A. J., Merritt, W. S., McKee, L. J., Eyre, B. D., and Baginska, B. (1999). Review of techniques to estimate catchment exports. Environmental Protection Authority, Sydney.

Letcher, R. A., Jakeman, A. J., Calfas, M., Linforth, S., Baginska, B., and Lawrence, I. (2002). A comparison of catchment water quality models and direct estimation techniques. *Environmental Modelling & Software* **17**, 77–85. doi:10.1016/S1364-8152(01)00054-8

Lewis, W. M., Melack, J. M., McDowell, W. H., McClain, M., and Richey, J. E. (1999). Nitrogen yields from undisturbed watersheds in the Americas. *Biogeochemistry* **46**, 149–162. doi:10.1007/BF01007577

Marston, F., Young, W., and Davis, R. (1995). Nutrient data book. Nutrient generation rates data book, 2nd edition. Unpublished CMSS Reference Manual. CSIRO Division of Water Resources, LWRRDC and SWB, Canberra.

McCaskill, M. R., Ridley, A. M., Okom, A., White, R. E., Andrew, M. H., Michalk, D. L., Melland, A., Johnston, W. H., and Murphy, S. R.

(2003). SGS Nutrient Theme: environmental assessment of nutrient application to extensive pastures in the high rainfall zone of southern Australia. *Australian Journal of Experimental Agriculture* **43**, 927–944. doi:10.1071/EA03020

McDowell, R. W., and Sharpley, A. N. (2001). Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of Environmental Quality* **30**, 508–520.

McDowell, R. W., Biggs, B. J. F., Sharpley, A. N., and Nguyen, L. (2004). Connecting phosphorus loss from agricultural landscapes to surface water quality. *Chemistry and Ecology* **20**, 1–40. doi:10.1080/02757540310001626092

McKergow, L. A., Weaver, D. M., Prosser, I. P., Grayson, R. B., and Reed, A. E. G. (2003). Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. *Journal of Hydrology* **270**, 253–272. doi:10.1016/S0022-1694(02)00286-X

McKergow, L. A., Prosser, I. P., Grayson, R. B., and Heiner, D. (2004). Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 2. Water quality. *Australian Journal of Soil Research* **42**, 485–498. doi:10.1071/SR02156

Merritt, W. S., Letcher, R. A., and Jakeman, A. J. (2003). A review of erosion and sediment transport models. *Environmental Modelling & Software* **18**, 761–799. doi:10.1016/S1364-8152(03)00078-1

Mitchell, A. W., Bramley, R. G. V., and Johnson, A. K. L. (1997). Export of nutrients and suspended sediment during a cyclone-mediated flood event in the Herbert River catchment, Australia. *Marine and Freshwater Research* **48**, 79–88. doi:10.1071/MF96021

Molloy, R., and Ellis, D. (2002). Don't just measure impacts, predict the changes we can achieve. In 'Coast to Coast, Tweed Heads, 4–8 November 2002'. pp. 295–298. (Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management: Brisbane.)

Monaghan, R. M., Paton, R. J., and Drewry, J. J. (2002). Nitrogen and phosphorus losses in mole and tile drainage from a cattle-grazed pasture in eastern Southland. *New Zealand Journal of Agricultural Research* **45**, 197–205.

Monaghan, R. M., Morton, J. D., McDowell, R. W., Drewry, J. J., and Thorrold, B. S. (2003). Livestock improvement corporation lecture: the development of environmental best practices for intensive dairying. *Proceedings of the New Zealand Society of Animal Production* **63**, 3–6.

Monaghan, R. M., Paton, R. J., Smith, L. C., Drewry, J. J., and Littlejohn, R. P. (2005). The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. *New Zealand Journal of Agricultural Research* **48**, 227–240.

Moore, I., and Gallant, J. (1991). Overview of hydrologic and water quality modelling. In 'Modelling the Fate of Chemicals in the Environment'. (Ed. I. Moore.) pp. 1–8. (Centre for Resource and Environmental Studies, The Australian National University: Canberra.)

Mundy, G. N., Nexhip, K. J., Austin, N. R., and Collins, M. D. (2003). The influence of cutting and grazing on phosphorus and nitrogen in irrigation runoff from perennial pasture. *Australian Journal of Soil Research* **41**, 675–685. doi:10.1071/SR02087

Nash, D., and Murdoch, C. (1997). Phosphorus in runoff from a fertile dairy pasture. *Australian Journal of Soil Research* **35**, 419–429. doi:10.1071/S96039

Nash, D., Halliwell, D., and Cox, J. (2002). Hydrological mobilization of pollutants at the field/slope scale. In 'Agriculture, Hydrology and Water Quality'. (Eds P. M. Haygarth and S. C. Jarvis.) pp. 225–242. (CABI Publishing: Wallingford.)

Nelson, P. N., Cotsaris, E., and Oades, J. M. (1996). Nitrogen, phosphorus, and organic carbon in streams draining two grazed catchments. *Journal of Environmental Quality* **25**, 1221–1229.

Newham, L. T. H., Cuddy, S. M., Vertessy, R. A., and Jakeman, A. J. (2004*a*). Informing the design of catchment modelling – a survey of end-user needs. CSIRO Land and Water and The Australian National University, Canberra.

Newham, L. T. H., Letcher, R. A., Jakeman, A. J., and Kobayashi, T. (2004*b*). A framework for integrated hydrologic, sediment and nutrient export modelling for catchment-scale management. *Environmental Modelling & Software* **19**, 1029–1038. doi:10.1016/J.ENVSOFT.2003.11.006

Newham, L. T. H., Cuddy, S. M., Vertessy, R. A., and Jakeman, A. J. (2005*a*). Determining future directions in contaminant cycle modelling through an evaluation of existing modelling systems. In 'SIMMOD. Simulation and Modelling: Integrating Sciences and Technology for Effective Resource Management, Bangkok, 17–19 January 2005'. (Eds V. Kachitvichanukul, U. Purintrapiban and P. Utayopas.) pp. 426–433. (Asian Institute of Technology: Bangkok.)

Newham, L. T. H., Rutherford, J. C., and Croke, B. F. W. (2005*b*). A conceptual model of particulate trapping in riparian buffers. Technical Report 21/05. CSIRO, Canberra.

Nexhip, K. J., and Austin, N. R. (1998). Defining key nutrient reduction practices for irrigated dairy pastures. In 'Proceedings of the Irrigation Australia 1998 National Conference and Exhibition: Water is Gold, Brisbane, 19–21 May 1998'. pp. 533–543. (Irrigation Association of Australia.)

Pakrou, N., and Dillon, P. (1995). Preferential flow, nitrogen transformations and 15N balance under urine-affected areas of irrigated and non-irrigated clover-based pastures. *Journal of Contaminant Hydrology* **20**, 329–347. doi:10.1016/0169-7722(95)00077-1

Pakrou, N., and Dillon, P. (2000). Key processes of the nitrogen cycle in an irrigated and a non-irrigated grazed pasture. *Plant and Soil* **224**, 231–250. doi:10.1023/A:1004838323594

Prosser, I., Rustomji, P., Young, B., Moran, C., and Hughes, A. (2001). Constructing River Basin Sediment Budgets for the National Land and Water Resources Audit. Technical Report 15/01. CSIRO Land and Water, Canberra.

Quinn, P. (2002). Models and monitoring: scaling-up cause-and-effect relationships in nutrient pollution to the catchment scale. In 'Agricultural Effects on Ground and Surface Waters: Research at the Edge of Science and Society. Proceedings of an International Symposium, Wageningen, Netherlands, October 2000'. (Eds J. Steenvoorden, F. Claessen and J. Willems.) pp. 397–403. (IAHS Press: Wallingford.)

Quinn, P. (2004). Scale appropriate modelling: representing cause-and-effect relationships in nitrate pollution at the catchment scale for the purpose of catchment scale planning. *Journal of Hydrology* **291**, 197–217. doi:10.1016/J.JHYDROL.2003.12.040

Rassam, D., Fellows, C., DeHayr, R., Hunter, H., Bloesch, P., and Beard, N. (2003). Interaction of surface water and groundwater and its potential effects on denitrification in riparian buffers. In 'Proceedings of the 28th International Hydrology and Water Resources Symposium, Wollongong, 10–13 November 2003'. (Ed. M. J. Boyd.) pp. 219–224. (The Institution of Engineers: Canberra.)

Ridley, A. M., White, R. E., Helyar, K. R., Morrison, G. R., Heng, L. K., and Fisher, R. (2001). Nitrate leaching loss under annual and perennial pastures with and without lime on a duplex (texture contrast) soil in humid southeastern Australia. *European Journal of Soil Science* **52**, 237–252. doi:10.1046/J.1365-2389.2001.00387.X

Ridley, A. M., Christy, B. P., White, R. E., McLean, T., and Green, R. (2003). North-east Victoria SGS National Experiment site: water and nutrient losses from grazing systems on contrasting soil types and levels of inputs. *Australian Journal of Experimental Agriculture* **43**, 799–815. doi:10.1071/EA02090

Robson, B. J., and Hamilton, D. P. (2003). Summer flow event induces a cyanobacterial bloom in a seasonal Western Australian estuary. *Marine and Freshwater Research* **54**, 139–151. doi:10.1071/MF02090

Rose, T., Donohue, R., and Robb, M. (2000). The West Australian experience with nitrogen: a review, opinions and identification of needs for better waterways management. In 'Nitrogen Workshop 2000: Sources, Transformations, Effects and Management on Nitrogen in Freshwater Ecosystems, Melbourne'. (Eds B. T. Hart and M. R. Grace.) pp. 60–64. Occasional Paper 10/00. Land and Water Australia, Canberra.

Rosich, R. S., and Cullen, P. (1982). Nutrient runoff. In 'Water Quality Management: Monitoring Programs and Diffuse Runoff'. (Ed. B. T. Hart.) pp. 103–119. (Water Studies Centre, Chisholm Institute of Technology and Australian Society for Limnology: Melbourne.)

Rutherford, K., Elliott, S., and Collins, R. (2003). Modelling nutrient runoff from agricultural land: how do we provide support for decision makers? In 'Tools for Nutrient and Pollutant Management: Applications to Agriculture and Environmental Quality'. (Eds L. D. Currie and J. A. Hanly.) pp. 312–319. Occasional Report No. 17. Fertiliser and Lime Research Centre, Massey University, Palmerston North.

Scholefield, D., Tyson, K. C., Garwood, E. A., Armstrong, A. C., Hawkins, J., and Stone, A. C. (1993). Nitrate leaching from grazed grassland lysimeters – effects of fertilizer input, field drainage, age of sward and patterns of weather. *Journal of Soil Science* **44**, 601–613. doi:10.1111/J.1365-2389.1993.TB02325.X

Sharpley, A. N., Weld, J. L., Beegle, D. B., Kleinman, P. J. A., Gburek, W. J., Moore, P. A., and Mullins, G. (2003). Development of phosphorus indices for nutrient management planning strategies in the United States. *Journal of Soil and Water Conservation* **58**, 137–151.

Silva, R. G., Cameron, K. C., Di, H. J., and Hendry, T. (1999). A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on nitrate leaching. *Australian Journal of Soil Research* **37**, 357–369. doi:10.1071/S98010

Smith, C., and Monaghan, R. M. (2003). Nitrogen and phosphorus losses in overland flow from a cattle grazed pasture in Southland. *New Zealand Journal of Agricultural Research* **46**, 225–237.

Stevens, D. P., Cox, J. W., and Chittleborough, D. J. (1999). Pathways of phosphorus, nitrogen, and carbon movement over and through texturally differentiated soils, South Australia. *Australian Journal of Soil Research* **37**, 679–693.

Swistock, B. R., Edwards, P. J., Wood, F., and Dewalle, D. R. (1997). Comparison of methods for calculating annual solute exports from six forested Appalachian watersheds. *Hydrological Processes* **11**, 655–669. doi:10.1002/(SICI)1099-1085(199706)11:7<655::AID-HYP525>3.0.CO;2-4

Toor, G. S., Condron, L. M., Di, H. J., Cameron, K. C., and Sims, J. T. (2004). Assessment of phosphorus leaching losses from a free draining grassland soil. *Nutrient Cycling in Agroecosystems* **69**, 167–184. doi:10.1023/B:FRES.0000029679.81951.BB

Tyson, K. C., Scholefield, D., Jarvis, S. C., and Stone, A. C. (1997). A comparison of animal output and nitrogen leaching losses recorded from drained fertilized grass and grass/clover pasture. *The Journal of Agricultural Science* **129**, 315–323. doi:10.1017/S0021859697004760

Vertessy, R. A., Watson, F. G. R., Rahman, J. M., Cuddy, S. M., Seaton, S. P., Chiew, F. H., Scanlon, P. J., Marston, F. M., Lymburner, L., Jeannelle, S., and Verbunt, M. (2001). New software to aid water quality management in the catchments and waterways of the south-east Queensland region. In 'Proceedings of the Third Stream Management Conference: The Value of Healthy Streams, Brisbane, 27–29 August 2001'. (Eds I. Rutherfurd, F. Sheldon, G. Brierley and C. Kenyon.) pp. 611–616. (Cooperative Research Centre for Catchment Hydrology: Canberra.)

Viney, N. R., and Sivapalan, M. (2001). Modelling catchment processes in the Swan-Avon river basin. *Hydrological Processes* **15**, 2671–2685. doi:10.1002/HYP.301

Viney, N. R., and Sivapalan, M. (2004). A framework for scaling of hydrologic conceptualizations based on a disaggregation-aggregation approach. *Hydrological Processes* **18**, 1395–1408. doi:10.1002/HYP.1419

Viney, N. R., Sivapalan, M., and Deeley, D. (2000). A conceptual model of nutrient mobilisation and transport applicable at large catchment scales. *Journal of Hydrology* **240**, 23–44. doi:10.1016/S0022-1694(00)00320-6

Wallbrink, P. J., Martin, C. E., and Wilson, C. J. (2003). Quantifying the contributions of sediment, sediment-P and fertiliser-P from forested, cultivated and pasture areas at the landuse and catchment scale using fallout radionuclides and geochemistry. *Soil & Tillage Research* **69**, 53–68. doi:10.1016/S0167-1987(02)00128-9

Wasson, R. J., Donnelly, T. H., and Murray, A. S. (1996). Imports can be dangerous – appropriate approaches to Australian rivers and catchments. In 'Proceedings of the First National Conference on Stream Management, Merrijig, 19–23 February 1996'. (Eds I. Rutherford and M. Walker.) pp. 313–319. (Cooperative Research Centre for Catchment Hydrology: Canberra.)

Weaver, D. M., and Reed, A. E. G. (1998). Patterns of nutrient status and fertiliser practice on soils of the south coast of Western Australia. *Agriculture Ecosystems & Environment* **67**, 37–53. doi:10.1016/S0167-8809(97)00096-0

Webster, I. T., and Harris, G. P. (2004). Anthropogenic impacts on the ecosystems of coastal lagoons: modelling fundamental biogeochemical processes and management implications. *Marine and Freshwater Research* **55**, 67–78. doi:10.1071/MF03068

Webster, I. T., Parslow, J. S., Grayson, R. B., Molloy, R. P., Andrewartha, J., Sakov, P., Tan, K. S., Walker, S. J., and Wallace, B. B. (2001). Gippsland Lakes environmental study. Assessing options for improving water quality and ecological function. Final report. CSIRO: Glen Osmond, Australia.

Whitmore, A. P., and Schroder, J. J. (1996). Modelling the change in soil organic C and N and the mineralisation of N from soil in response to applications of slurry manure. *Plant and Soil* **184**, 185–194. doi:10.1007/BF00010448

Wilcock, R. J., Nagels, J. W., Rodda, H. J., O'Connor, M. B., Thorrold, B. S., and Barnett, J. W. (1999). Water quality of a lowland stream in a New Zealand dairy farming catchment. *New Zealand Journal of Marine and Freshwater Research* **33**, 683–696.

Young, W. J., Marston, F. M., and Davis, J. R. (1996). Nutrient exports and land use in Australian catchments. *Journal of Environmental Management* **47**, 165–183. doi:10.1006/JEMA.1996.0043

Manuscript received 5 September 2005; revised 1 May 2006; and accepted 6 September 2006.