Technical report

Review of Techniques to Estimate Catchment Exports

EPA NSW

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PREFACE

This Technical Report was commissioned by the NSW Environment Protection Authority as part of its National Pollutant Inventory (NPI) Program funded by Environment Australia. The NPI is a database designed to provide the community, industry and government with information on the types and quantities of pollutants emitted to air, land and water environments.

This Report—*Review of Techniques to Estimate Catchment Exports*—focuses on the estimation of nutrient emissions from diffuse sources to waters. It consolidates a contemporary understanding of generic catchment water quality models and estimation techniques. A comprehensive listing of methods and estimation techniques is provided together with information to identify their relative merits. The aim of this review was to identify 'best practices' approaches for NPI assessments of aggregated catchment emissions to water but it is anticipated that it will also be relevant to a much broader audience.

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EXECUTIVE SUMMARY

Reliable estimation and prediction of diffuse pollutant exports on a catchment scale are fundamental to an understanding of aquatic systems and thus are critical to the development of strategies to limit impacts in our waterways. A plethora of models and techniques exist to estimate the export of nutrients from catchments. The selection of an appropriate estimation technique requires clearly defined objectives, knowledge of available data and an understanding of the strengths and weaknesses of various approaches.

This report provides:

- an overview and categorisation of catchment models including details of commonly used models with respect to their assumptions, inputs required, complexity, ease of use, availability and application to Australian catchments
- a review of model acceptance criteria and the uncertainty associated with model output
- a review and demonstration of existing methods for pollutant load estimation based on direct observation
- an inventory of nutrient generation rates
- an inventory of modelling groups in Australia.

It is concluded that physics based models and the more complex conceptual models are not particularly appropriate for estimating catchment exports across most Australian catchments for the following reasons:

(i) lack of sufficient spatially distributed input data to drive the models

(ii) paucity of calibration data in space and time to define an appropriate parameter set for the models and hence reliable output

(iii) the over-dependency of model results on the experience of the user

(iv) for physics based models in particular, demanding computational requirements at large catchment scales.

On the other hand, empirical and conceptual approaches can be combined constructively to provide models without these problems and with the following properties:

- event responsiveness and sensitivity to climate variability
- allow investigation of catchment source strengths
- general physical interpretability of modelling results.

Empirical and simple conceptual models are more appropriate for estimating catchment exports across most Australian catchments than physics based or complex conceptual models.

It is also concluded that there is no single optimal sediment and nutrient (direct) load estimation technique. Selection of an appropriate load estimation technique depends not only on the availability of concentration and discharge data, but also on the hydrological characteristics of the catchment being considered, the desired accuracy of estimates and the preferred complexity of the load estimation technique. All techniques considered were found to have disadvantages in certain situations.

A number of load estimation methods were tested using a data set from the Richmond River catchment, northern NSW. When nutrient and suspended sediment data exist that cover the entire annual flow regime for a year (flow weighted or stratified collection), several methods gave accurate estimates of nutrient and sediment loads. In some situations, routine data (such as weekly, monthly or seasonal collection) may also give an accurate estimate of annual loads. Firstly, if by chance several samples during routine collection are collected during a high flow event, then a rating curve may be developed. Secondly, routine data may be suitable in regulated catchments where flow does not change greatly over the annual cycle. Daily turbidity data collected for water supply purposes is a special case when routine data may be extremely reliable for the assessment of pollutant loads after it is calibrated. For instance, turbidity data sets, which exist throughout NSW, could be retrieved and calibrated for nutrient and suspended sediment concentration.

It is recommended that for data that is collected for calculating pollutant loads, the following best practices be adopted:

- any future pollutant concentration data should be collected using a flow weighted or stratified approach that has a bias towards periods of high flow when concentrations are highly variable and when the majority of loads are transported
- the data should be collected at or close to a flow gauging point
- methods such as linear interpolation, interval concentration, interval discharge, or flow-weighted average concentration should be used to calculate the load.

In catchments where data does not allow determination of pollutant loads, a number of empirical methods give reasonable load estimates and some of these will allow ranking between catchments. Multi-factor methods (e.g. Moss *et al.* 1993) are recommended as an appropriate method for load estimation when no data exists. These methods are likely to give better predictions than simple computer models, which rely on average sediment and nutrient generation rates for different land uses obtained from the literature, when their coefficients for runoff, erosion, sediment delivery, nutrient enrichment, and dissolved load compensation are known for similar catchments. In such cases, empirical methods are likely to provide better long-term average (versus event) estimates than simple computer models at a fraction of the cost. It is recommended that a specific model using this multi-factor approach be developed using existing data sets. Such a model will complement empirical computer modelling, allow a relative ranking of catchments, evaluate the effects of each land use, and evaluate the significance of urban and fertiliser inputs.

These load estimation and multi-factor methods complement each other as well as complementing the role of conceptual models. Indeed, as argued in section 4.2.8, the three approaches can be combined constructively to provide results which are more productive than the sum of the results of the three methods used in isolation from one another. Conceptual models allow exports to be sensitive to climatic events and antecedent conditions before an event, and hence can provide an estimate of the variability of exports. The multi-factor method can be used to parameterise, or predict the effects of spatial variability of catchment exports where no measurements exist. The type of conceptual model chosen should be as accurate and parametrically efficient as possible while incorporating a description of the key processes such as quick flow component, instream advection, suspension and resettling. Components of a number of simple conceptual models, LASCAM-WQ, IHACRES and STARS (see section 3.2), can be utilised to achieve these two objectives. An approach combining a simple conceptual model framework with an empirical technique, such as the multi-factor method, could provide useful information on water quality in data-poor situations, where more complicated conceptual and physics-based models are inappropriate, due to the lack of available input data, and the difficulty in identifying parameter values.

The report reviews a number of important areas of the literature, as well as providing examples of sediment load estimation techniques using a data set from the Richmond River catchment. Chapter 1 provides an introduction to issues associated with sediment and nutrient export from catchments, as well as outlining the major sections of the report. Sediment and nutrient load estimation techniques are reviewed in Chapter 2, with a number of techniques being demonstrated and compared on a test data set. Chapters 3 and 4 present a review of issues associated with sediment and nutrient transport modelling, beginning with a general review of model classification types and calibration criteria, moving to a review of specific models available for sediment and nutrient transport modelling. A description of specific models mentioned throughout the report is given in section 3.2. Features of specific models are also summarised in Table 4.1, at the end of Chapter 4. Appendix I gives a review of the literature on nutrient generation rates, which is supported by individual reference summaries. A list of modelling groups within Australia is provided in Appendix II.

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1 INTRODUCTION

Human influences on the landscape such as vegetation removal, introduction of exotic species, fertilisation, and the development of urban areas constitute drastic modifications of both nutrient cycles and fluvial sedimentary regimes. In an Australian context, there is evidence of radical changes in sediment and nutrient dynamics. For example, nutrient and sediment loads in Queensland coastal rivers may have increased by 3–5 times since pre-European settlement (Moss *et al.* 1993) and estimates for the Richmond River catchment range from a 2 to 5 times increase for nitrate, phosphate and suspended sediment loads (Eyre 1997; Hossain 1998). Eutrophication problems such as those found in the Peel–Harvey Estuary of Western Australia or the Murray–Darling Basin have raised awareness within the community about the impact of land use on rivers and waterways.

Much research has focused on the development and application of modelling techniques for the prediction or estimation of water quality responses to changes in land use or land management practices and on the impact of rainfall and topographical features on water quality. In urban areas, the changes in water quality in response to stormwater inflows and sewerage overflows into the catchment system have also been the focus of considerable research (USEPA 1983; Simeoni *et al.* 1994). For example, recent years have seen an increase in the use of empirical models of flow in geomorphology and hydrology (Lane 1998). In addition, conceptual (e.g. LASCAM, IHACRES-STARS) and physics based (e.g. WEPP) models have been used. The use of models can act to further our understanding of key processes in sediment and nutrient generation and can be used as a management tool for community groups at a catchment or subcatchment level, modelling the impact of changes in catchment management policy.

The modelling of water quality in catchment systems requires an understanding of the processes involved. For suspended solids these processes include soil erosion, sediment transport and sediment deposition. However, many models fail to account for all three processes. For example, USLE is an erosion prediction model that does not account for sediment deposition (Zhang *et al.* 1995). These processes are linked with one another, to the extent that it is difficult to discuss one in isolation from the others. Sediment transport and the flow of water are an example of this, where the two processes are so closely linked that it is hard to discuss sediment yield modelling without considering modelling of flow (Bennett 1974). In this introduction, the key processes in erosion, sediment transport and deposition, as well as nutrient export from catchments, will be discussed as they relate to water quality modelling, with particular reference to Australian conditions.

1.1 Sedimentary processes

Increased sediment exports from catchments have been identified as the cause of channel incision (Wasson 1998), increased downstream sedimentation (Finlayson 1996; Hossain

1998), and the transport of pollutants (Finlayson 1996). Catchment sediment processes can be categorised into erosion, delivery, and export. The eventual export of sediment out of a catchment is a function of many interacting processes. There is uncertainty associated with random components such as rainfall, antecedent soil moisture, and soil cover. In general, sediment erosion increases as rainfall intensity and slope increase and as vegetation cover decreases (Finlayson 1996). Rainfall droplets hit the catchment surface, breaking up larger soil particles as well as providing a flotation medium. Slope determines the velocity of runoff, which directly affects the sediment detachment. Soil structure, texture, and composition also help to determine soil erosivity which, together with the soil roughness, also affects the velocity of the runoff, which determines entrainment of sediment (Novotny 1989). Vegetation cover reduces soil erosion by reducing the impact velocity of rain droplets and reducing runoff through interception, evapotranspiration and the binding of the soil together by plant roots.

Sediment eroded from catchment surfaces and stream channels may either be redeposited within the catchment system or be exported from the catchment as fluvial sediment load. The amount of sediment transported from the catchment may be an order of magnitude less than the amount of soil erosion (Novotny 1989). This is caused by 'flow competency' either in-channel or during overland flow. If at any time during sediment transport the carrying capacity of the flow is exceeded by sediment supply, then excess sediment will be deposited. This illustrates a start–stop motion typical of sediment transport. It has been suggested that sediment spends more time in storage than in transport (Meade 1982). The ratio of eroded sediment carried by a stream outlet from a catchment to the on-site erosion within the catchment is termed 'the delivery ratio'. Although convenient in concept, the idea of a delivery ratio has been severely criticised because of the way it spatially and temporally averages a given catchment area. Sediment erosion and transport operate on a wide variety of time scales including diurnal and seasonal. High intra- and interannual variability in Australian rainfall and runoff are likely to enhance the discontinuous nature of erosion and delivery in Australian catchments, thus highlighting the question of how important catastrophic events are for pollutant export (Webb and Walling 1982) and therefore what time frame is valid for averaging. The delivery ratio has also been criticised for spatially averaging a given catchment area. For instance, a catchment area that is disturbed and eroding rapidly but spatially removed (either by distance or by an obstacle) from a stream may deliver less sediment to the stream than an area of low erosion potential in close proximity to the stream (Novotny and Chesters 1989). Wasson (1996) suggested that the link between water quality and land use varies with position in the catchment, because of variation in connectivity between hill slopes, flood plains and channels. These concepts may be of great importance when modelling other pollutants associated with sediment (Novotny 1989).

There is conflict as to whether increased catchment erosion has actually resulted in increased sediment yield. There is evidence in some Australian catchments that the majority of post-European sediment erosion is still stored within the catchment (Brizga and Finlayson 1994; Rutherford and Smith 1992; Grayson *et al.* 1994). In contrast, erosion of soils following the extension of the sugar cane industry in the Johnson River catchment, Queensland, correlates well with downstream sediment accumulation

(Conner 1986). Larger catchments usually have lower stream slope, lower water velocity, and wider flood discharge peaks. As a result, catchment size appears to strongly influence the sediment delivery. Further, the signature of catchment disturbance also decreases with an increase in the catchment size (CSIRO 1992). However, this may not be the case in urban areas (Novotny 1989). In a recent summary, it was suggested that there are several major differences between Australian catchments and the rest of the world. Sheet and rill erosion on an area-weighted basis are higher than the global average, the sediment discharge to the oceans is lower than the global average, and the sediment delivery ratio (< 3%) is much lower than the global average (Wasson 1996). This will potentially affect the selection and use of overseas techniques for sediment load estimates and modelling packages in Australian conditions.

1.2 Nutrients

Nitrogen occurs naturally in catchment soils, being fixed from the atmosphere by both symbiotic and non-symbiotic microbes in soils, associated with plant roots, and on the surfaces of plant leaves and stems (Attiwill 1987). Nitrogen is additionally washed out of the atmosphere in rainfall (Correll 1982; Meybeck 1982; Hinga 1991). Nitrogen content in rocks is very low (Meybeck 1982). Nitrogen sources from erosion of the parent rock are usually considered unimportant. In contrast, phosphorus is common in igneous rocks and, unlike nitrogen, most naturally occurring phosphorus in catchments is ultimately derived from the weathering of parent substrate (Attiwill 1987; Wasson 1996). Australian soils are typically poor in phosphorus and this is mainly associated with age and subsequent long leaching periods (Young 1996). In natural systems it is likely that the majority of nitrogen and phosphorus is conserved with little leaching, as evidenced by relatively low nutrient exports from forested areas (Young 1996).

Anthropogenic activities within catchments have highly modified catchment nutrient cycles through the introduction of domestic animals and addition of nitrogen and phosphorus fertilisers to pastures and crops (Figure 1.1) (CSIRO 1992). For example, organic manures derived from intensive animal husbandry are reused to enhance crop production, while wastes and runoff from urban areas and factories are discharged to catchment waterways. An estimate of the Australian phosphorus budget (Figure 1.1) indicates that inputs exceed losses by 298 kt yr^{-1}(CSIRO 1992) The nitrogen budget for Australia does not include estimates of denitrification and volatilisation and thus the estimate for nitrogen fixation is considered a minimum value (Figure 1.1). As such, there is no estimate for soil nitrogen enrichment, however there is substantial evidence suggesting that nitrogen storage in a system increases in proportion to nitrogen inputs (Frissel 1978) and therefore it is likely that Australian catchments are also accumulating nitrogen.

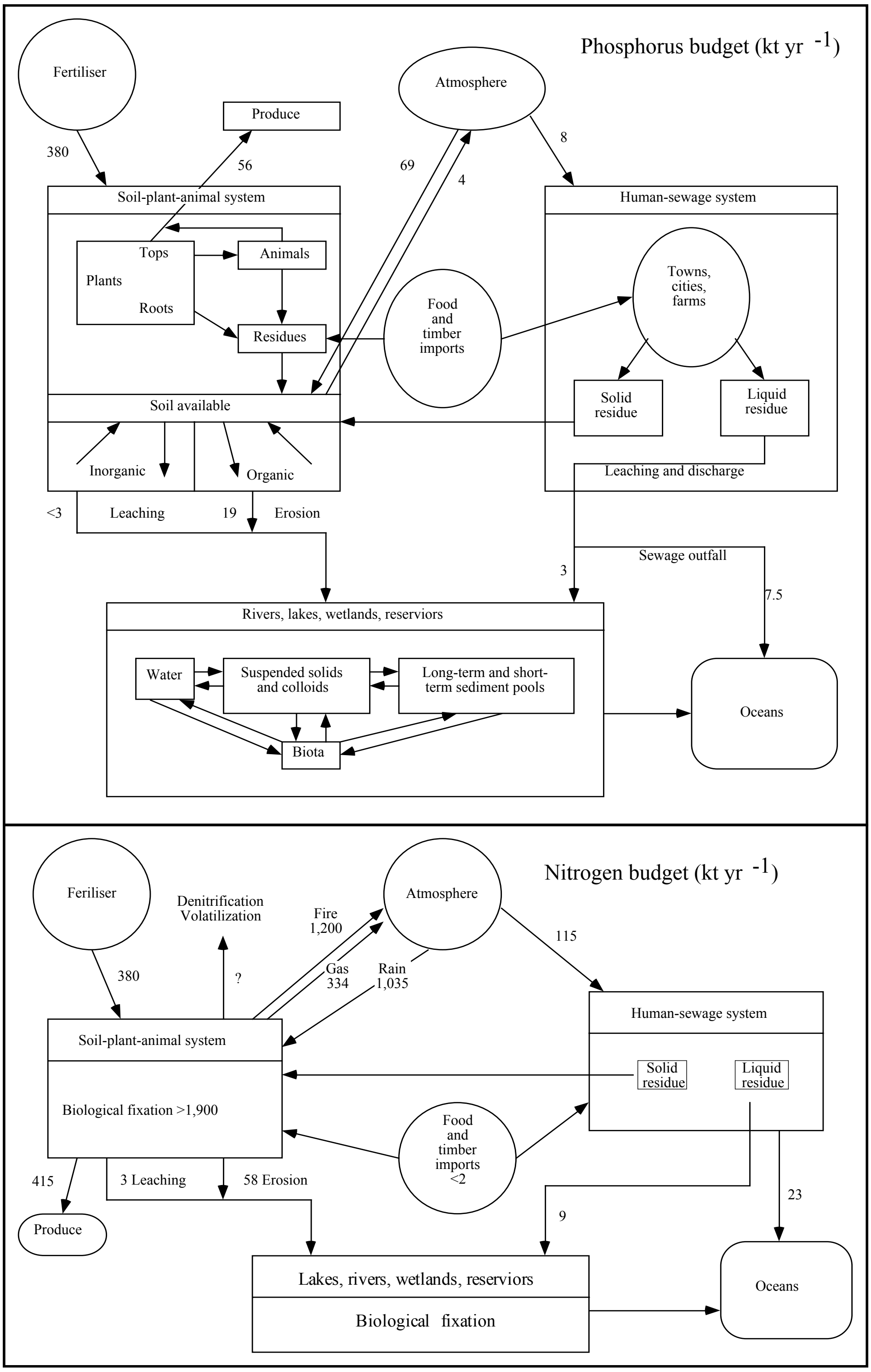

Figure 1.1: Estimated nutrient budgets for Australia (CSIRO 1992)

Nutrients stored in or supplied to catchments may be leached or eroded during storm events and subsequently transported from the catchment surfaces to the drainage network. Meybeck (1982) summarised this, suggesting that suspended pollutants can be derived from three origins: erosion and dissolution of inorganic nutrients, leaching of inorganic nutrients derived from mineralisation of terrestrial organic matter, and leaching and erosion of organic soil components. The dominant pathway that a nutrient follows is catchment specific and depends on the nutrient source, catchment slope, rainfall and runoff relationships, soil properties, the affinity of the nutrient to soils, rates of biological uptake, and the nutrient form. For instance, phosphorus has a high affinity to soil particles and was found to be tightly bound to soils on the Darling Plateau in Western Australia (Gerritse 1995). As such, the phosphorus exported off catchment surfaces is often only a small percentage of phosphorus fertiliser inputs (Sharpley 1987). The transport of phosphorus is usually associated with soil particles, and particulate phosphorus can be up to 77% of the total phosphorus loads in rivers (e.g. Cosser 1989; McKee 1996). Therefore, changes in catchment sedimentary cycles also have implications for the transport of phosphorus.

The majority of nitrogen transported in world rivers is in dissolved organic forms (Meybeck 1982) and, on a catchment scale, nitrate is recognised as highly leachable. As a result, a proportion of nitrogen added to crops and pastures may be leached to waterways. Despite differences in chemical affinity, the majority of both nitrogen and phosphorus loads are transported in catchments during flood discharge. For instance 86% of the annual total phosphorus load in the South Pine River catchment, south-eastern Queensland, is transported in 2.8% of the time. Knowledge of flow paths may be important for nutrient management on a farm scale, whereby nutrient losses are reduced by avoiding fertilisation during seasons with high runoff or by using slow-release fertilisers that are less likely to leach to groundwater. Such knowledge will also be important for the nutrient component of water quality modelling as this will affect the relationships that are used in modelling.

1.3 Sediment and nutrient load estimation techniques

There are several techniques available for the estimation of sediment and nutrient loads, falling into two main categories: real data and empirical methods. Real data methods include a variety of averaging, ratio and regression methods. These methods can be used when streamflow and nutrient/sediment load concentration measurements are available. Empirical methods are used to estimate loads when there is an absence of such observed data. The nutrient or sediment load for a river can be predicted using relationships between sediment and nutrients and other readily available environmental attributes, such as population. Chapter 2 contains a discussion of a number of the methods available, both real data and empirical methods, including an application of many of the techniques to estimating nutrient and sediment loads in a Richmond river ‘test’ catchment.

1.4 Erosion and sediment/nutrient transport modelling

Computer models of erosion and sediment/nutrient transport fall into three broad categories: empirical models, conceptual models and physics-based models. Empirical models are generally based on simple stochastic or empirically determined relationships found between observed variables. Conceptual models are based on the conceptualisation of the catchment as a configuration of internal storages and pathways. Physical relationships are not considered explicitly but are represented in general terms through the conceptualisation of the catchment. Physics-based models are grounded on the solution of fundamental physical equations of flow and transport. Each of these classes of models has a number of advantages and disadvantages. Many models are not clearly definable as belonging to any one category, but possess a combination of components from different classes. Also, the categories used to classify models are not universally agreed upon by the modelling community. A number of different classification groups, as well as definitions of model types, may be found in the literature. The best model will depend on a number of factors, including the intended use of the model, the data and computing resources available and the expertise of the model user. Chapter 3 includes a discussion of these general model types. A detailed outline of many specific erosion and sediment/nutrient transport models, including model equations and a review of Australian applications, is given in section 3.2. A review of calibration acceptance criteria and discussion of the predictive capacity of models are provided in Chapter 4.

1.5 Nutrient generation rates

Nutrient generation rates depend on factors such as land use and management, soil type, topography, climate, and antecedent conditions. This means that nutrient generation rates reported for catchments outside Australia, such as in the North American literature, are likely to be different from those that can be expected in Australia. However, the Australian literature on nutrient generation rates is fairly sparse, and thus is often augmented with data from North American studies (Young *et al.* 1996). Appendix I provides a review of Australian literature on nutrient generation rates and summaries of specific references on nutrient generation rates.

2 LOAD ESTIMATION TECHNIQUES

2.1 Load estimation using field data

Estimation of the load of suspended sediment and other pollutants is an important part of analysing the response of a catchment to rainfall events. Loads are not generally measured directly in-stream; rather, load estimates are inferred from measurements of pollutant concentration and water discharge in-stream.

In general, pollutant load, L, over a time period, T, can be represented by the equation

$$\mathrm{L} = \int_0^T CQdt \tag{2.1}$$

where C is the pollutant concentration and Q is the water discharge.

A close approximation to this load equation is given by

$$\mathrm{L} = \sum_{i=1}^{\frac{T}{\delta t}} C_i Q_i \tag{2.2}$$

where the sampling interval, δt, is short compared to the period of time over which the discharge and concentration vary. Most techniques for pollutant load estimation are based on this equation involving concentration and water discharge. In practice, this equation is usually not able to be used directly to calculate pollutant loads, as the sampling period for discharge and/or concentration is longer than the period over which concentration and discharge are invariant. However, when the sampling interval approaches the concentration variability with flow, the method of linear interpolation can be used to generate 'real' nutrient and sediment loads. As such, linear interpolation has been used in studies which test the accuracy and bias of the other methods (e.g. Young and DePinto 1988; Kronvang and Bruhn 1996). Linear interpolation can be described by the following equation (Kronvang and Bruhn 1996):

$$\sum_{i=1}^{n+1} \sum_{t_i < t \le t_{i+1}}^{i} q_t \frac{C_{t_i}(t_{i+1} - t) + C_{t_{i+1}}(t - t_i)}{t_{i+1} - t_i} \tag{2.3}$$

where concentrations are denoted C_{ti}, t_i, $i = 1,...,n$ are the times at which concentration is measured, t_0 and t_{n+1} are the times at the start and end of each subinterval, and q_t is the discharge for each time step.

Unfortunately, although river discharge is usually sampled frequently, often at intervals of less than a day, and mostly continuously, pollutant concentrations are generally sampled infrequently, often at routine intervals (i.e. daily, weekly, monthly, or seasonally). In these cases, linear interpolation may be inappropriate, and calculating pollutant loads usually involves another method of estimating pollutant concentrations for the period within the sampling interval. Generally, less confidence may be placed on loads estimated using data with a wide sampling interval or a sampling interval that does not characterise flood events. The ideal sampling interval will vary from one river to another. Accurately capturing variations in sediment and nutrient load given the rapid fluctuations in sediment and nutrient concentration is a major problem in some rivers. This issue is particularly important in small catchments, where response to rainfall events tends to be rapid, and in subtropical Australian catchments, where 50% of the annual discharge can occur in 3% of the time (e.g. Cosser 1989). In larger rivers, such as the Richmond River, northern NSW, a sampling interval of four times a day characterises the concentration variability adequately, whereas in a steep mountain stream, the water level may rise and fall over a 24 hour period and a pollutant sampling interval of less than one hour may be more appropriate.

There are many different techniques used for calculating load estimates, differing in complexity, accuracy and bias. The choice of technique may depend on the data resolution, the mathematical ability of the operator, the computer technology available, or the relationships within the data and between various pollutant concentrations. Ideally, data should be collected to suit a particular river and a particular method of load estimation. However, more often data are collected without clear objectives thus reducing collection efficiency and usefulness.

2.1.1 Methods

2.1.1.1 Averaging

Averaging methods are generally considered to be the simplest available techniques for pollutant load estimation, and are often applied because of a lack of more appropriate techniques. Estimates of load over a time period are made by using averages of discharge, concentration or load for a given subinterval and then summing these over the entire period. These averages may be over different time periods, such as monthly, quarterly or yearly, and can combine discharge and concentration in a number of different ways (Table 2.1). Whilst these methods are easy to apply, the assumptions implicit behind such calculations, including independent and identically distributed data, are rarely met. This leads to bias in the estimation of loads, especially if the sampling program does not collect data from the entire range of discharge and concentration variability. Where a positive relationship occurs between concentration and discharge, loads will tend to be underestimated by time averaging, and where a negative relationship exists, loads will usually be overestimated. The magnitude of over- or underestimation will depend on the range in variation in concentration (Walling and Webb 1985). This effect is likely to be more severe for suspended sediments which tend to show stronger positive relationships with discharge than nutrients which are transported in both dissolved and particulate form (Walling and Webb 1985).

Walling and Webb (1981) considered a number of average estimators in a comparison of the precision and accuracy of load estimation procedures on a river in the United Kingdom using empirical techniques. They found that Methods 1, 3 and 6 (Table 2.1), underestimated load by 70% or more. Methods that weight concentration by discharge at the time of sampling (Methods 2, 4 and 5) are more likely to produce accurate load estimates. However, the precision of these methods, as indicated by the standard deviation of load estimates produced, is generally less than the precision for Methods 1 and 3. Walling and Webb (1981) conclude that Methods 1 and 3 produce the most worthwhile results, the consistency of which indicates that the use of a correction factor may be appropriate with these methods. At the very least these methods would be most likely to reproduce approximately the relative ranking of pollutant load. All six estimation methods that Walling and Webb (1981) considered showed a sharp drop in the precision of load estimates as the sampling interval increased.

Similar conclusions were made by Preston *et al.* (1989). They also found that average estimators using daily discharge with monthly or quarterly average concentration frequently have the lowest mean squared error. Such estimators were found to have a high precision, however, their accuracy was sometimes poor. Other average estimators were found to have a higher bias or variance or both. Estimators using average discharge data were found to be inaccurate and imprecise especially under event sampling. The precision of average estimators was found to slightly improve when the calculations were stratified.

Clarke (1990) also considered the mean and variance of loads calculated using Methods 1, 2, and 3 (Table 2.1). Clarke (1990) made the assumption that suspended sediment concentration and mean daily discharge are bivariate, log-normally distributed. Clarke (1990) found that Method 2 provided an unbiased estimate of suspended sediment load, whereas the estimate provided by Method 1 was negatively biased with respect to that provided by Method 2. However, the variance associated with Method 2 has variance of the order $1/n$ whereas the variance associated with Method 1 is of the order $1/n^2$. These results support and explain the empirical results found by Walling and Webb (1981).

Table 2.1: Averaging techniques for the determination of annual riverine loads

Method	Load Equation	Source
1	$k\left(\sum_{i=1}^{n}\frac{c_i}{n}\right)\left(\sum_{i=1}^{n}\frac{q_i}{n}\right)$	Walling and Webb (1981)
2	$k\left(\sum_{i=1}^{n}\frac{c_i q_i}{n}\right)$	Walling and Webb (1981)
3	$k\overline{q}\left(\sum_{i=1}^{n}\frac{c_i}{n}\right)$	Walling and Webb (1981)
4	$k\frac{\sum_{i=1}^{n}c_i q_i}{\sum_{i=1}^{n}q_i}\overline{q}$	Walling and Webb (1981)
5	$k\sum_{i=1}^{n}c_i\overline{q}_{p_i}$ where p_i denotes the period between samples	Walling and Webb (1981)
6	$k\sum_{i=1}^{12}\overline{c}_m\overline{q}_m$ where m denotes the month	Walling and Webb (1981)
7	$Q\sum_{i=1}^{n}\frac{c_i}{n}$	This study
8	$\sum_{j=1}^{n}\frac{c_j + c_{j+1}}{2}q_j$	Lesack (1993)
9	$\sum_{m=1}^{12}\sum_{j=1}^{N_m}q_{jm}\left[\sum_{i=1}^{n_m}\frac{c_{ijm}}{n_m}\right]$	Preston *et al.* (1989)
10	$\sum_{h=1}^{4}\sum_{j=1}^{N_h}q_{jh}\left[\sum_{i=1}^{n_h}\frac{c_{ijh}}{n_h}\right]$	Preston *et al.* (1989)
11	$\frac{365}{n}\sum_{i=1}^{n}q_i c_i$	Preston *et al.* (1989)
12	$\frac{365}{12}\sum_{m=1}^{12}\left[\frac{\sum_i q_{im}}{N_m}\right]\left[\frac{\sum_i c_{im}}{n_m}\right]$	Preston *et al.* (1989)
13	$\frac{365}{n}\sum_{h=1}^{4}\left[\frac{\sum_i q_{ij}}{N_h}\right]\left[\frac{\sum_i c_{ih}}{n_h}\right]$	Preston *et al.* (1989)
14	$\sum_{k=1}^{2}\frac{N_k}{n_k}\left[\sum_{i=1}^{n_k}q_{ik}c_{ik}\right]$	Preston *et al.* (1989)

Notation:
n = number of days sampled
N = total number of days
Q = total discharge
$\overline{q}$ = average discharge
k = scaling factor for the length of the period considered
q_j = discharge during the sampling interval j

2.1.1.2 Ratio estimators

Ratio estimators aim to take advantage of correlation within a sample. Generally discharge data is used as an auxiliary variable, x_i, with load data treated as a dependent variable, y_i. The ratio estimate is usually calculated as

$$Y_R = (y/x)\ X \qquad (2.4)$$

where y and x are the sample means of y_i and x_i respectively, Y_R is the ratio estimate of load and X is the discharge.

If y_i/x_i is nearly the same for all sampling units, y/x varies little from one sample to another and the ratio estimate is of high precision.

The ratio estimator is the best linear unbiased estimator under two conditions:
1. the relationship between x_i and y_i is a straight line passing through the origin
2. the variance of y_i about the line is proportional to x_i.

In general these conditions will not be met, so that the ratio estimator is biased, although consistent. Preston *et al.* (1989) found that the ratio estimators they considered (Table 2.2) were more often less precise than other approaches considered, but were virtually unbiased in each test case. This most likely reflected that the underlying distributions of the data considered for each test case were appropriate for ratio estimation.

Preston *et al.* (1989) observed little difference between ratio estimates under non-event scenarios, although they found the simple ratio method to be slightly more precise. In all cases stratification under event sampling was found to virtually eliminate bias, however stratification under non-event sampling did not improve estimation and was often found to reduce precision slightly. Stratification is the term used to describe separating the data into groups with common attributes (e.g. high flow, rising stage, falling stage, event, low flow, seasonal etc.). Overall, ratio estimators were found to be more robust than other estimation methods, virtually unbiased in all test cases, but slightly less precise than the averaging and regression methods that were tested.

Table 2.2: Ratio estimators for the determination of annual riverine loads

Method	Load Equation	Source
15	$\frac{\bar{l}}{\bar{q}}Q$	Preston *et al.* (1989)
16	$\bar{r}Q + \frac{n(N-1)}{(n-1)}(\bar{l} - \overline{rq})$	Preston *et al.* (1989)
17	$\hat{R}_Q Q$	Preston *et al.* (1989)
18	$\left[\hat{R}_{-} + \frac{n(N-n+1)}{N\bar{q}}(\bar{l} - \hat{R} - \bar{q})\right]Q$	Preston *et al.* (1989)
19	$\hat{R}\left[1 - \left(\frac{1}{n} - \frac{1}{N}\right)\left(\frac{s_q^2}{\bar{q}^2} - \frac{s_{lq}}{\bar{l}\bar{q}}\right)\right]Q$	Preston *et al.* (1989)
20	$\hat{R}\left[\frac{1 + \left(\frac{1}{n} - \frac{1}{N}\right)\left(\frac{s_{lq}}{\bar{l}\bar{q}}\right)}{1 + \left(\frac{1}{n} - \frac{1}{N}\right)\left(\frac{s_q^2}{\bar{q}^2}\right)}\right]Q$	Preston *et al.* (1989)

Notation:
n = number of days sampled
N = total number of days
Q = total discharge
$\bar{l}$ = average load
$\bar{q}$ = average discharge
$\bar{r}$ = average sample ratio
$\hat{R}_Q$ = jackknifed ratio
R = estimated ratio over population
s_{lq} = covariance between load and flow
s_q^2 = variance of flow
Note:
1. For the average discharge and average load variables, the period over which this should be calculated has not been specified by the authors. However, it is generally considered that they should be daily averages calculated over a ‘sufficiently’ long time period to capture a wide range of behaviour.
2. The jackknifed ratio is calculated using the equation

$$\hat{R}_Q = \frac{1}{n}\sum_{j=1}^{n}\left(nR - (n-1)R_j\right)$$

where R_j is calculated by removing the jth observation from the population and estimating the ratio over this subset of the population.

2.1.1.3 Regression Estimators

Regression estimators, also commonly referred to as rating curves, have been widely applied to estimating suspended sediment loads. Regression estimators are based on extrapolating a limited number of concentration measurements over the entire period of interest by developing a relationship between pollutant concentration or load and stream discharge, and applying this relationship to the entire discharge record. Typically this relationship is considered to be log–log, that is, the log of pollutant load or concentration is assumed to be a linear relationship of the log of stream discharge. This relationship is generally applied because both discharge and concentration are often best described by a bivariate log–normal distribution.

It has been found in a number of studies that the regression curve estimates based on this log–log relationship are biased, systematically under-predicting sediment loads. The reasons for this bias have been analysed and discussed by a number of authors (Preston *et al. 19*89; Ferguson 1986, 1987; Singh and Durgunoglu 1989). Essentially, whilst parameters are unbiased in log–log space, the process of transforming parameter estimates back to the stream discharge or stream load space introduces bias into the parameter estimates. This means that the rating curve that is calculated underestimates the pollutant yield. Ferguson (1986) found that ratings curves can under-predict by up to 50% of the load value, even when a full concentration time series is available. The degree to which the ratings curve underestimates load values increases with the amount of scatter of the data around the ratings curve. Care must be taken to ensure that rating curves are not used in inappropriate situations; for instance, where the relationship between discharge and concentration is not log–log or when a small number of observations are used and the relationship between discharge and concentration is not clearly revealed for a range of conditions (Preston *et al.* 1989). Further, discharge concentration relationships with time often follow loop relationships either on an event basis (due to hysteresis for example) (Walling and Webb 1980) or on a seasonal basis (Davis and Keller 1983). To improve accuracy in these situations it is usually necessary to stratify the data and develop two or more rating relationships depending on the complexity of the data.

A number of solutions to the problem of bias have been presented, generally based on the concept of a bias correction factor. One such correction factor was suggested by Ferguson (1986), where the correction factor varied with the mean squared error of the log-transformed regression, given as Method 21 (Table 2.3). Cohn *et al.* (1989) criticise this correction factor, stating that it does not eliminate bias, and that it can lead to severe overestimation of loads. Cohn *et al.* (1989) compare a traditional ratings curve to two modified curve structures, including that given as Method 22 (Table 2.3). They state that this method is unbiased and performs nearly as well as or better than the other approaches in all cases, assuming that the hypothesis of the log–linear model is correct. Findings presented by Walling and Webb (1988) indicate that the use of correction factors for log–log relationships did not lead to more accurate estimates of sediment load.

Walling and Webb (1981) highlighted major discrepancies between load estimates using ratings curve methods. For example, two studies conducted in New Zealand (Griffiths

1979; Adams 1980) using rating curve regression methods showed a consistent discrepancy. On average, the results of Adams (1980) were 70% higher than those of Griffiths (1979); results on one particular river differed by nearly two orders of magnitude. In a two-year study of an 85.5 km^2 rural Hunter Valley catchment, Loughran (1977) developed rating relationships from data stratified seasonally and for rising and falling stage. There were no discernible seasonal patterns revealed, however there were distinct variations between rising and falling stage. Sediment loads were calculated using two methods: rating relationships and measured concentration integrated with hourly flow. There were large discrepancies found between the loads calculated by each method, however when the two years were considered together, the methods agreed within 8%.

Table 2.3: Regression techniques for the determination of annual riverine loads

Method	Load Equation	Source
21	$\sum_{i=1}^{365} q_i \left[\exp\left(\hat{b}_0 + \hat{b}_1 \ln q_i\right)\right] \exp\left(\frac{s_{cr}^2}{2}\right)$	Ferguson (1986)
22	$\sum_{i=1}^{365} q_i \left[\exp\left(\hat{b}_0 + \hat{b}_1 \ln q_i\right)\right] MVUEBCF$	Preston *et al.* (1989)
23	$\sum_{i=1}^{365} q_i \left[\exp\left(\hat{b}_0^* + \hat{b}_1^* \ln q_i\right)\right] \exp\left(\frac{s_{cr}^2}{2}\right)$	Preston *et al.* (1989)
24	$\sum_{i=1}^{365} q_i \left[\exp\left(\hat{b}_0 + \hat{b}_1 \ln q_i\right)\right]$	Kronvang and Bruhn (1996)

Notation:
a,b = parameters estimated by regression of log-transformed concentration and discharge
q = discharge
s_{lq} = covariance between load and discharge
s_{cr}^2 = variance of residual in prediction of concentration
MVUEBCF = minimum variance unbiased estimator bias correction factor
$\hat{b}_o, \hat{b}_1$ = fitted regression parameters
$\hat{b}_o^*, \hat{b}_1^*$ = fitted robust regression parameters

2.1.1.4 Discussion

Preston *et al.* (1989) completed a comparison of three classes of load estimation techniques. The classes they investigated were simple averaging methods, ratio estimation methods and regression methods. They evaluated a number of different techniques within these classes using Monte Carlo sampling studies. Preston *et al.* (1989) found that no group of estimators were better in all cases. In general it was found that ratio estimators, whilst often imprecise, were virtually unbiased in all test cases. Ratio estimators also appeared to be more robust than regression estimators to bias caused by many hydrological or constituent characteristics. Differences between groups of estimators were found to be generally due to violations of model assumptions caused by test case characteristics. Unfortunately, ratio techniques tend to require a greater number of samples to achieve the same precision as regression estimators. Since estimators within each category were designed for data meeting specific statistical criteria, then an appropriate load estimator would be available if the characteristics of the data were well defined (Preston *et al.* 1989). In practice, limited data and information means that it is generally difficult to identify the most appropriate method for load estimation. Successful application may depend on the development of an *a priori* scheme for identifying whether an estimator is appropriate (Preston *et al.* 1989).

Preston *et al.* (1989) also considered stratified versions of all ratio and regression estimator methods. Stratification is a modification of the approaches presented. The population is divided into homogeneous subunits called strata which are sampled separately, and the estimates are combined to obtain an estimate over the entire population. It was found that stratification could substantially improve the accuracy and precision of estimates. The results found by Preston *et al.* (1989) indicate that this is not always the case. Whereas the precision of averaging techniques improved slightly with stratification, error levels were found to be higher when stratified sampling is used with regression techniques.

The selection of an appropriate load estimation technique therefore depends not only on the availability of concentration and discharge data, but also on the hydrological characteristics of the catchment being considered, the desired accuracy of estimates and the preferred complexity of the load estimation technique. No single technique has been found to be optimal in the literature, with all techniques having some disadvantages associated with their use. The choice of technique will depend on the characteristics of the catchment being considered, and the availability of data for that catchment.

2.1.2 Results

As outlined, there are many methods for calculating nutrient and sediment loads (Tables 2.1, 2.2, 2.3) due to both the sampling strategy and the difficulty of integrating continuous streamflow data with non-continuous concentration data. Each method employs a different strategy for approximating the concentration between discrete samples, so each is likely to give a different load with respect to the same period of time. In this section some of the methods outlined above will be used to calculate loads using a 'test' data set for a one-year period.

2.1.2.1 Test data set

The subcatchment chosen covers an area of 1790 km^2 above the township of Casino in the subtropical Richmond River catchment, northern NSW. The catchment has mixed rural land use (beef and dairy grazing, forest for timber, and a small amount of urban and crop lands). A stratified sampling approach was used during data collection in the Richmond River catchment. Samples were collected routinely on a monthly basis and up to six times per day during flood discharge. Water samples for nutrient analysis were taken during the rising and falling stages of the flood hydrograph. Samples were taken from approximately mid-depth at three points across the stream using a sample-rinsed submersible bottle and placed in a sample-rinsed 10 L bucket. Subsamples were taken from the bucket and placed in acid-washed and sample-rinsed 10 mL polyethylene vials. Samples for suspended sediment analysis were filtered through preweighed 0.45 μm filters using a vacuum of 30 kPa. The filter paper was placed in a vial and the volume of filtrate was recorded. Samples were placed on ice until frozen in the laboratory freezer (–20°C) within twelve hours. Laboratory analysis was carried out of a Lachat Instruments Auto Analyser using cadmium reduction (nitrogen) and molybdate blue (phosphorus) following persulphate digestion. Suspended sediments were measured by weighing each filter paper after drying to constant mass at 60°C (approximately 8 hours) and subtracting the mass of the filter paper with sediment from the mass of the filter paper without sediment.

In total, 70 water samples were collected between July 1995 and June 1996 and 74% of the samples were collected during three floods of varying magnitude (Figure 2.1). The NSW Department of Land and Water Conservation records stage height at Casino, and discharge is calculated using stage-discharge rating relationships. For the purpose of this study, discharge was retrieved on an hourly and a daily basis. Turbidity data (Figure 2.1) is collected routinely 365 days a year by Casino Shire Council for the purposes of assessing treatment requirements for town water supply. Water is drawn from the Richmond River 3 km upstream of Casino above a 3 m weir, 1 m from the bottom, and 8 m from the river bank. Turbidity is measured at source at about 12:00 noon daily using a Great Lakes Instruments turbidity meter Model 8202. Comparisons are regularly made using a Hach Model 2100A.

Nutrient and sediment loads for the test catchment were generated using linear interpolation on a one-hour time step (Table 2.4). These are believed to be accurate and unbiased estimates of nitrogen, phosphorus and suspended sediment loads, and all other methods considered will be compared to the loads generated using linear interpolation.

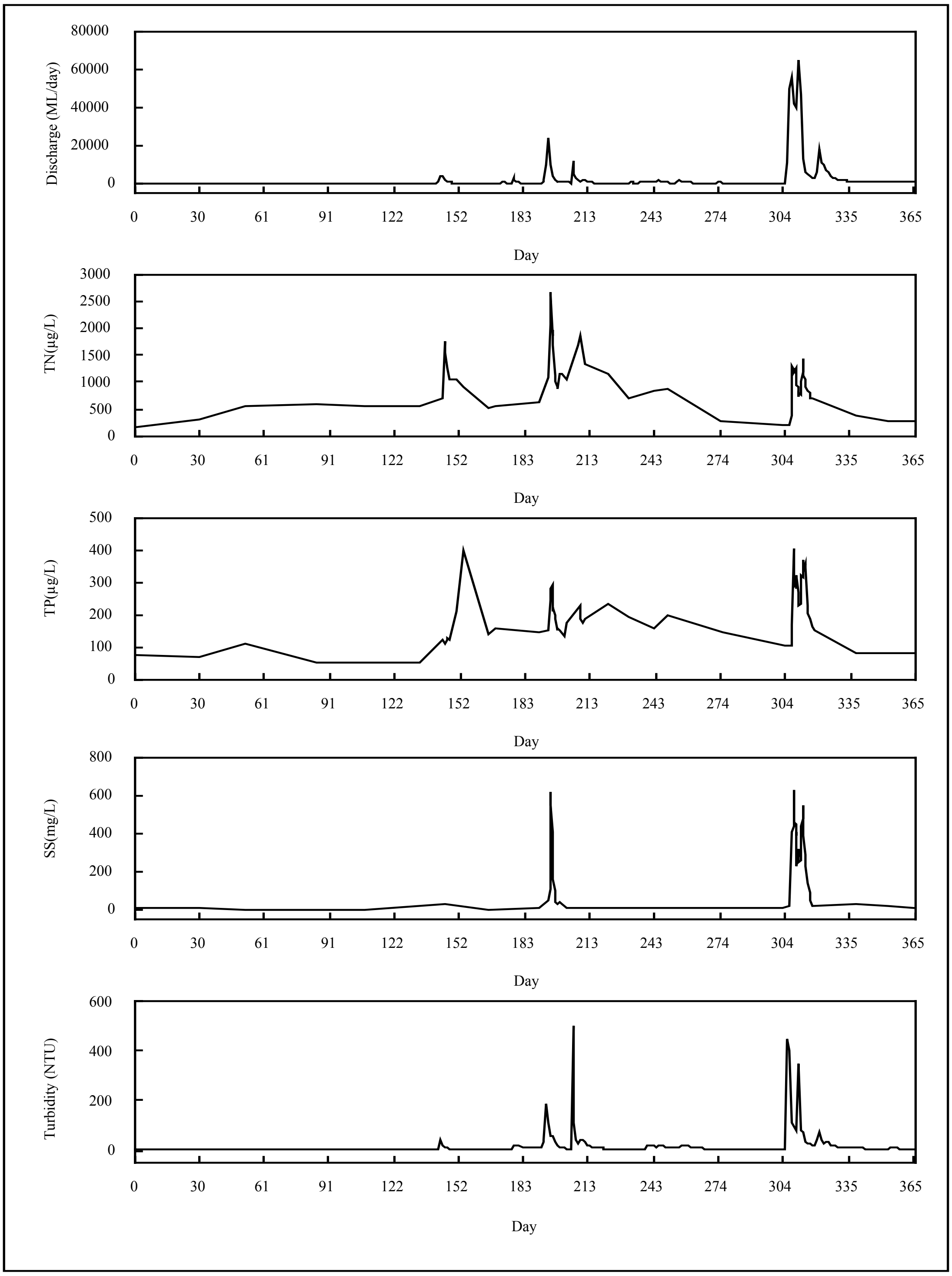

Figure 2.1: Discharge, total nitrogen (TN), total phosphorus (TP), suspended sediment (SS), and turbidity data for a subcatchment of the Richmond River catchment hereafter denoted the test catchment

Table 2.4: Comparison of the nutrient and sediment export in the test catchment for a range of mathematical methods

(The factor difference is calculated as the load of the method in question divided by the load estimated by linear interpolation)

Method	Description	Total N (t)	Factor difference	Total P (t)	Factor difference	SS* (10^3 t)	Factor difference
Real	Linear interpolation	606	–	142	–	141	–
1	Average sample concentration × average sample discharge scale for time	6709	11.1	1343	9.50	1389	9.90
1a	Same as 1 only scaled for discharge	620	1.02	124	0.87	128	0.90
2	Average sample load scaled using time	7722	12.70	1883	13.30	3182	22.60
2a	Same as 2 only scaled using discharge	714	1.18	174	1.23	294	2.10
7	Arithmetic mean × annual discharge	620	1.02	124	0.87	128	0.91
8	Concentration for a sample interval × discharge for the sample interval	597	0.99	144	1.01	142	1.01
15	Flow weighted concentration × annual discharge	714	1.18	174	1.23	223	1.58
15a	Stratified version of method 13	682	1.13	160	1.13	194	1.38
24	Log space discharge-concentration regression	663	1.09	143	1.01	138	0.98
24a	Log space discharge-load regression	663	1.09	143	1.01	138	0.98

* SS = suspended sediment

2.1.2.2 Averaging methods

Table 2.1; Methods 1, 2

In methods 1 and 2, actual mean flow is approximated by the mean of the flow for the limited number of samples. This approximated flow is then used to calculate load, and the sum of the sampled load is scaled up to an annual load by a time factor. In the test catchment there were 70 nutrient and 53 sediment samples; therefore the factors were 8784 hours / 70 hours and 8784 hours / 53 hours. Methods 1 and 2 overestimated nutrient and sediment loads by between 9.5 and 22.6 × (Table 2.4) because the average of the sampled discharge was 11 × greater than the actual mean hourly discharge for the year 1995–96. When the loads were scaled using discharge rather than time (Table 2.4; Methods 1a, 2a) a close agreement was found with the real loads except for suspended sediment (2a), which had a positive bias of 2.1 ×. It should be noted that Methods 1a and 2a are equivalent to Method 7 and Method 15. The improvement in load estimation using discharge rather than time to scale the sampled loads was expected, since the sampling program was stratified to capture flood events and therefore was biased towards high flow. If the sampling regime had not been biased towards high flow and floods had not been sampled so rigorously, it is likely that Methods 1 and 2 would have underestimated pollutant load in the Richmond River catchment, as discharge would then have been greatly underestimated. Methods 1 and 2 are unlikely to give accurate results when river flow varies substantially, however, in regulated rivers where dams control discharge, they may be suitable.

2.1.2.3 Arithmetic mean

Table 2.1; Method 7

The accuracy and bias of this method reflects how well the concentration data set represents the 'true' range of concentrations. Time averaging is normally thought of as one of the least reliable methods of calculating nutrient and sediment loads in streams. For example, combining mean concentration with annual flow produced a negative bias for all test cases when calculating sediment loads using routinely collected (weekly, fortnightly, or monthly) data (Walling and Webb 1985). In the Richmond River test catchment, nitrogen loads were slightly overestimated (1.02 ×), phosphorus loads were underestimated (0.87 ×) and suspended sediment loads were slightly underestimated (0.91 ×) using this method (Table 2.4). Apparently the arithmetic mean of the stratified data collected in the test catchment approximates the theoretical mean generated by linear interpolation.

2.1.2.4 Interval concentration and discharge

Table 2.1; Method 8

Concentration for the interval between samples is approximated by averaging the sample concentration at the start and end of the interval. The average concentration is then combined with the discharge for the interval, and each interval load is added to give the annual total. The loads generated by this method are within 1% of those calculated by linear interpolation for both nutrients and suspended sediment (Table 2.4). This close agreement results from relatively small changes in nutrient and suspended sediment concentrations between samples (on average < 20% change) during flood discharge in the Richmond River test catchment and is a direct result of an adequate sampling interval.

2.1.2.5 Flow-weighted mean concentration

Table 2.2; Method 15

This simple ratio method integrates flow and sample concentration as a means of averaging the concentration data. The accuracy of the method will depend on how representative the sample distribution is across the range of flow regimes. In rivers which are event driven, where very high flows occur for days during wet season floods and very low flows occur for the rest of the year, the outcome will usually be negatively biased if samples are collected on a routine basis (weekly or monthly).

In the test catchment, where samples were collected on a stratified basis, the outcome appears to be positively biased by between 18% and 58% (Table 2.4). In this case, 74%

of the samples were collected during three floods when nutrient and sediment concentrations were high. The positive bias was greater for sediments than for nutrients when compared to the linear interpolation method ('Real') because sediment concentration is more closely aligned with flow and varied by 224 × over the test year. Nutrient concentrations remain elevated for a longer period during flood flow and may be elevated during post-flood conditions due to nutrient-rich groundwater. During the dry season, nutrient concentrations may also be elevated due to the influence of point sources. As a result, nutrient concentrations varied only by a factor of 16 for nitrogen and 8 for phosphorus over the one year test period. The greater bias supports Walling and Webb (1985), who suggested that the degree of over- or underestimation will depend on the magnitude of variation of the sample concentration.

A reduction of the positive bias can be seen when the flow-weighted averaging was carried out on a monthly basis, thereby singling out flood periods which happened to fall within calendar months (Table 2.4; Method 15a). A further improvement is likely if the data set had been stratified into rising stage, falling stage, post-flood flow and low-flow months. Stratification, either on a flow or temporal basis, is often found to reduce bias (e.g. Kronvang and Bruhn 1996). Stratification can also be achieved on an event basis using event mean concentration and event discharge. This could be incorporated with a separate load calculation for low-flow months either monthly or seasonally depending on the available data. Indeed, there are many ways of stratifying data and stratification often results in a better approximation of the real loads.

2.1.2.6 Regression

Table 2.3; Method 24

Two common regression techniques have been tested. Relationships were developed for nitrogen, phosphorus and suspended sediment in log space (Figure 2.2). Concentration data were calculated using an hourly time step and integrated with hourly flow. Hourly loads were then summed to give a load for the year. The process was repeated for load–discharge relationships, and the loads thus derived on an hourly time step were summed to give a load for the year. The concentration and load methods gave a similar result (Table 2.4). With respect to linear interpolation, regression methods slightly overestimated nitrogen load (1.09 ×) and slightly underestimated suspended sediment loads (0.98 ×). These results suggest that the number and distribution of the samples over the range of discharges accurately characterised the positive relationships between discharge and pollutant concentration. The bias correction factors suggested by a number of authors (e.g. Ferguson 1986; Ferguson 1987; Walling and Webb 1988) appear to be unnecessary for the test catchment.

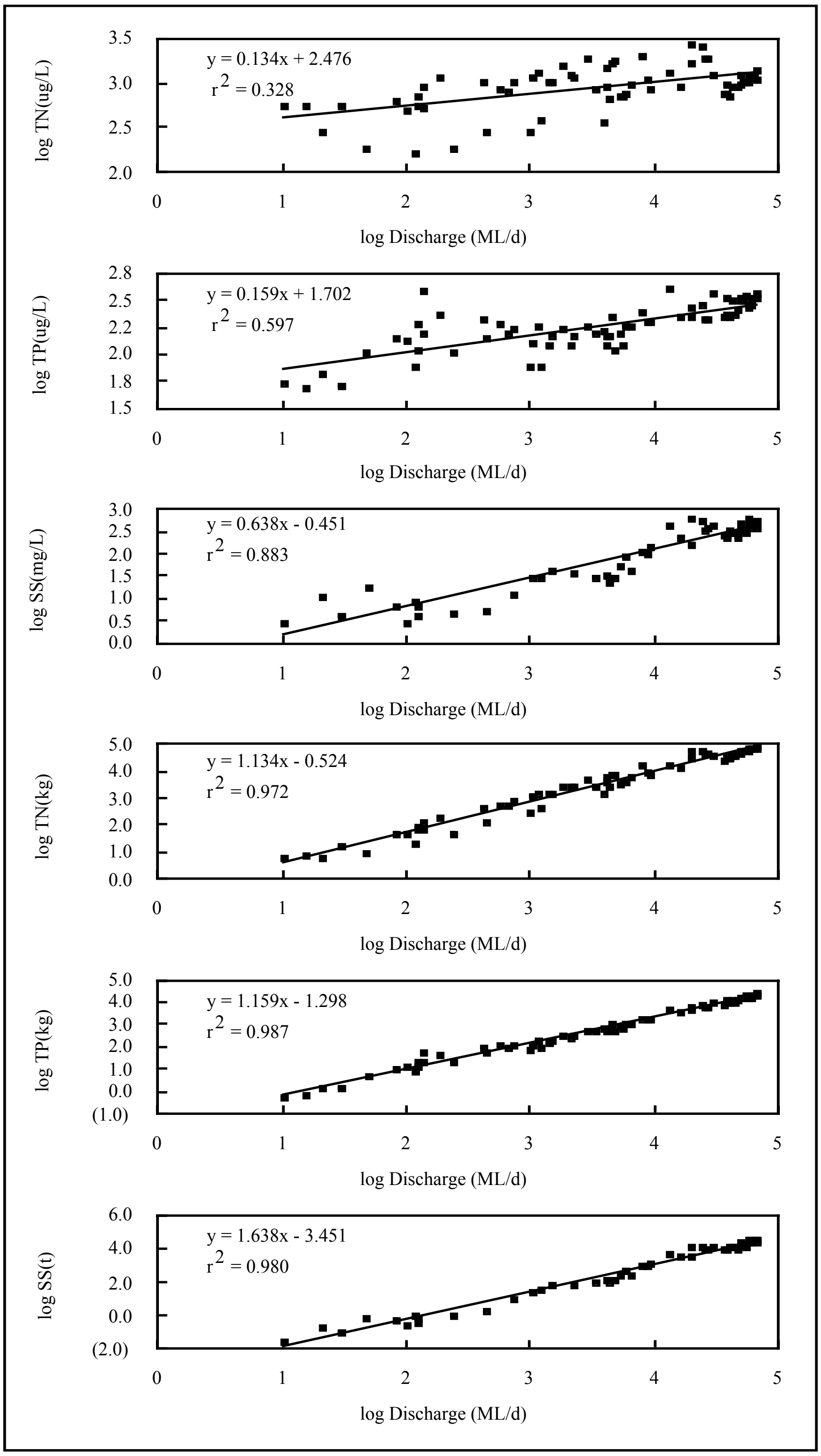

Figure 2.2: Regression analysis of the Richmond River test data in log–log space

2.2 Empirical models

In the absence of field-collected data, the nutrient or sediment load in a river draining to a receiving water body can be predicted by relationships generated using data from other catchments either nearby or in other parts of the world. This section tests the predictive ability of relationships between nitrogen, phosphorus and suspended sediments and other readily available environmental attributes. The discussion will be limited to total nitrogen, total phosphorus, and suspended sediments, however, similar relationships may be derived for other nutrient forms (e.g. nitrate, phosphate) or contaminants. Each empirical method will be used to estimate the sediment and nutrient loads for the Richmond River test catchment and will be compared to the loads generated by linear interpolation.

2.2.1 Methods

2.2.1.1 Population density as a predictor of nutrient loads

The presence of a human population in a catchment causes disturbance that leads to greater nutrient inputs to the catchment as well as greater release of nutrients stored within vegetation and soils. Several studies have shown significant relationships between population and nitrate-nitrogen and phosphate-phosphorus in 42 major rivers of the world (Cole *et al.* 1993; Caraco 1995). These studies are important in that they highlight the role that humans play in ecosystems and imply that water bodies adjacent to areas of greater population density (such as the NSW coast) are likely to be affected by greater nutrient loads. Simple regression relationships (Figure 2.3) were developed using data collected from published scientific literature on population density and total nitrogen loads (Yarbro *et al.* 1984; Billen *et al.* 1985; Cooper and Thomsen 1988; Boynton *et al.* 1995; Howarth *et al.* 1996; Walling *et al.* 1997; McMahan and Woodside 1997; Freifelder *et al.* 1998), and population density and total phosphorus loads (Dillon and Kirchner 1975; Yarbro *et al.* 1984; Pilleboue and Dorioz 1986; Cooper and Thomsen 1988; Cosser 1989; Kronvang 1992; Boynton *et al.* 1995; Howarth *et al.* 1996; McMahan and Woodside 1997; Walling *et al.* 1997; Dorioz *et al.* 1998). Virtually all the data were from temperate catchments, the exceptions being the Richmond River catchment in northern NSW and the South Pine River catchment in south-eastern Queensland.

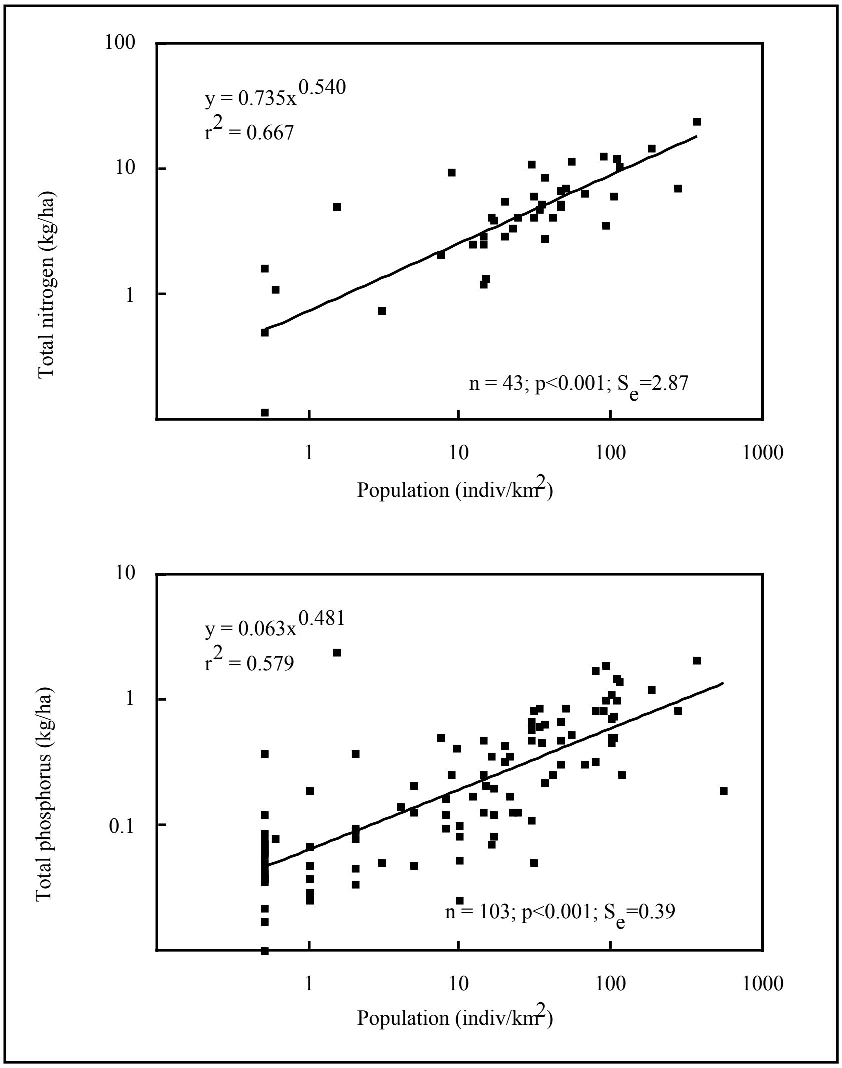

Figure 2.3: The relationships between the population density in a catchment and nutrient export in rivers and streams

2.2.1.2 Fertiliser addition as a predictor of nutrient export

Excessive use, methods of application, and the timing of the application of fertilisers in rural catchments are often implicated as contributing to eutrophication of adjacent water bodies (Lukatelich *et al.* 1987). As such, it seems likely that the fertiliser loading in a catchment may correlate with riverine nutrient export (e.g. Birch 1982). To test if this hypothesis holds between catchments, data were gathered from published scientific literature on fertiliser application and total nitrogen exports (Alberts *et al.* 1978; Frissel 1978; Groth *et al.* 1978; Beaulac and Reckhow 1982; Billen *et al.* 1985; Jaworski *et al.* 1992; Howarth *et al.* 1996; Nelson *et al.* 1996; Hoyas *et al.* 1997; Jordan *et al.* 1997; McMahan and Woodside 1997; Freifelder *et al.* 1998) and fertiliser loading and total phosphorus exports (Alberts *et al.* 1978; Beaulac and Reckhow 1982; Birch 1982; Lowrance *et al.* 1985; Pilleboue and Dorioz 1986; Sharpley and Menzel 1987; Jaworski *et al.* 1992; Nelson *et al.* 1996; McMahan and Woodside 1997; Dorioz *et al.* 1998). With the exception of data from Western Australia and the Richmond River system in northern NSW there are few data available in published scientific literature on Australian systems. The relationships generated (Figure 2.4) had relatively low correlation coefficients ($r^2 < 0.3$) but were significant ($p < 0.001$).

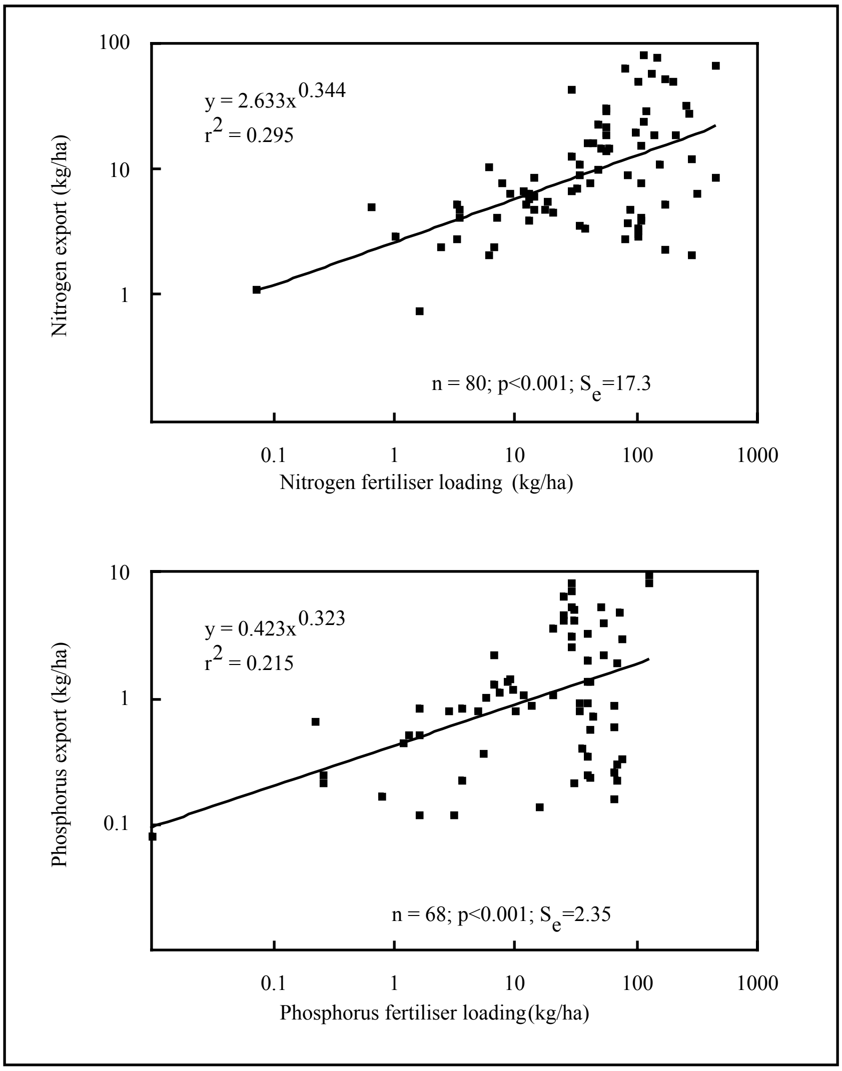

Figure 2.4: The relationships between catchment fertiliser use and nutrient export in rivers and streams

2.2.1.3 Relationships between nutrients and suspended sediment

A proportion of nitrogen and phosphorus in river systems is transported in association with inorganic and organic soil particles. For example, 77% of phosphorus export in the South Pine River, south-eastern Queensland, is particulate phosphorus and 29% of nitrogen exported in the Richmond River is transported as particulate organic nitrogen (Cosser 1989; McKee and Eyre 1996). Many authors have reported relationships between particulate matter and nitrogen (e.g. Meybeck 1982) and phosphorus (Kronvang 1992).

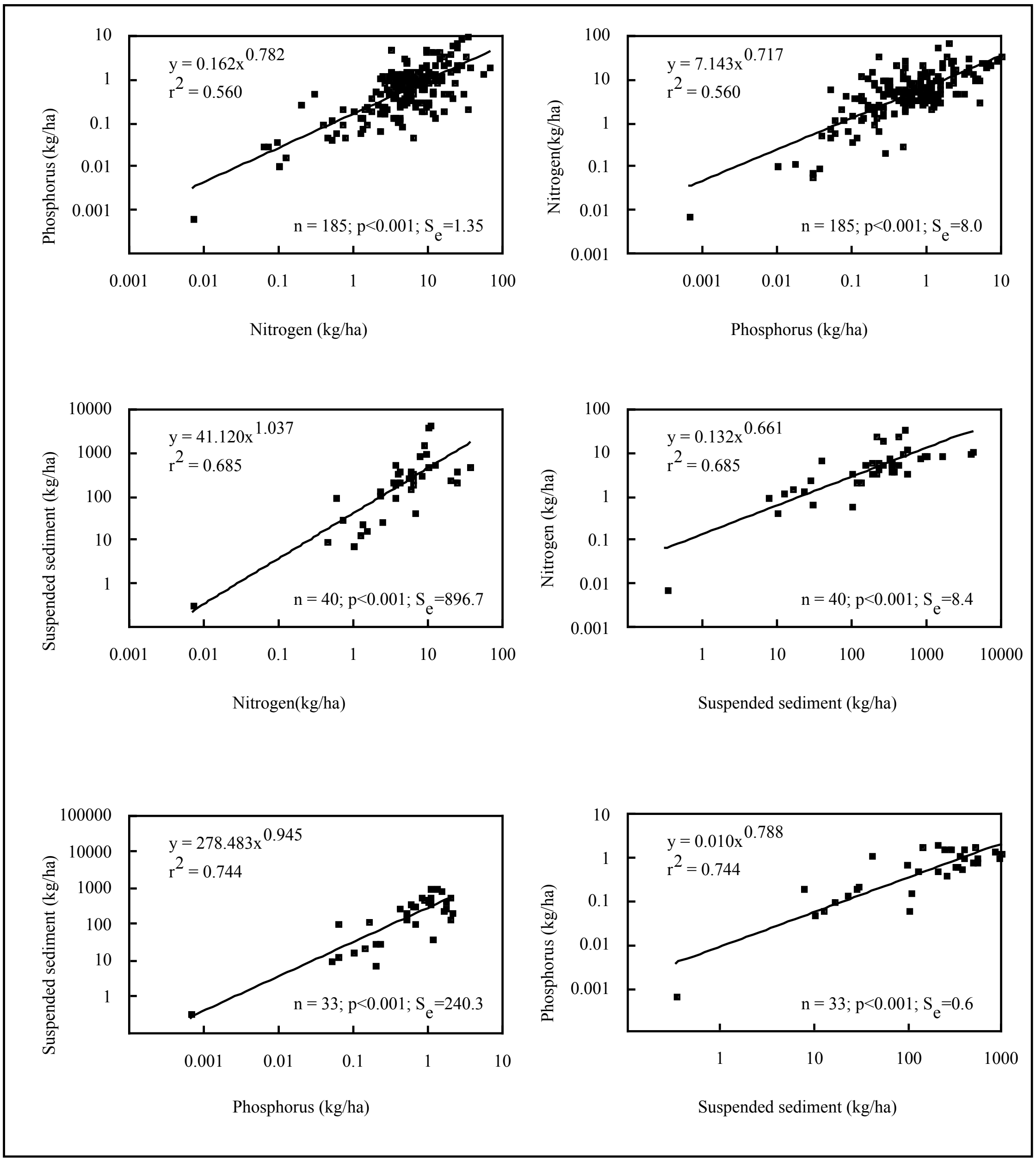

Figure 2.5: The relationships between the export of nitrogen, phosphorus, and suspended sediment exports in rivers

Regression models were developed using data collected from published scientific literature on catchments where nitrogen and phosphorus export were known (Alberts *et al.* 1978; Correll 1981; Meyer *et al.* 1981; Beaulac and Reckhow 1982; Hoare 1982; Lesack *et al.* 1984; Yarbro *et al.* 1984; Lowrance *et al.* 1985; Cooper and Thomsen 1988; Frink 1991; Jaworski *et al.* 1992; Correll *et al.* 1992; Hunter 1993; Boynton *et al.* 1995; Howarth *et al.* 1996; Nelson *et al.* 1996; McMahan and Woodside 1997; Jordan *et al.* 1997; Mitchell *et al.* 1997; Walling *et al.* 1997), where suspended sediment and nitrogen exports were known (Hunter 1993; McDowell and Asbury 1994; Nelson *et al.* 1996; Mitchell *et al.* 1997; Walling *et al.* 1997), and where suspended sediment and phosphorus exports were known (Pedrozo and Bonetto 1987; Kronvang 1992; Hunter 1993; McDowell and Asbury 1994; Nelson *et al.* 1996; Mitchell *et al.* 1997; Walling *et al.* 1997). The relationships generated (Figure 2.5) were all highly significant ($P < 0.001$).

2.2.1.4 Land use and sediment and nutrient exports

Land used for different purposes may be 'disturbed' to differing degrees depending on, for example, tillage practices, fertiliser application rates and timing, stocking densities, urbanisation, and industrialisation. Although there is much variation, average nutrient exports appear to be horticulture > cropland > urban > improved pasture > pasture > forest (Beaulac and Reckhow 1982; Frink 1991; Young *et al.* 1996). For sediment export, the order (on average) is overgrazed pasture > crop land > native pasture > undisturbed forest (e.g. Neil and Fogarty 1991; Hill and Peart 1998). An estimate of total nutrient and sediment export from a catchment area can be made if the area of each broad land use category in known and a nutrient or sediment generation rate is applied. This concept has been developed by CSIRO into the computer modelling package CMSS and has also been used in catchments in other parts of the world (e.g. Haith and Shoemaker 1987; Mattikalli and Richards 1996). The following nutrient export rates (Table 2.5) were suggested for tropical Australian catchments (Young *et al.* 1996).

Table 2.5: Nutrient generation rates (kg/ha/yr) used to estimate nutrient exports from the Richmond River test catchment

Land use	Nitrogen	Range	Phosphorus	Range	Suspended sediment	Range
Cropping	12.3	–	1.9	–	570	420–4000
Horticulture	26.0	20–34.5	7.1	2.7–14.3	420?	420–4000
Grazing (fertilised)	5.0	0.6–10.8	1.1	0.1–1.9	420?	420–1000
Grazing	3.0	2.2–5.1	0.1	0.002–0.4	190	140–1000
Urban	6.6	1.0–22.4	1.0	0.1–3.6	160	160–1000
Forest	0.9	0.9–1.5	0.1	0.001–0.2	40	20–60

* Note that the upper values in this table were used to calculate nutrient and sediment loads in the test catchment.

Sediment export rates (kg/ha/yr) are based on Neil and Fogarty (1991) and Wasson *et al.* (1996) and on outputs from a spatially and temporally calibrated HSPF computer model developed for the Johnstone River catchment, Queensland, discussed in Appendix I.

2.2.1.5 Sediment yield and nutrients found in catchment soils

An estimate for nutrient export rate may be made by multiplying the sediment yield of a particular catchment by the concentration of nutrients in catchment soils. This method was used for calculating phosphorus export from small catchments (<10 km^2) on the Southern Tablelands of NSW and the ACT (Wasson *et al.* 1996). In this case Wasson *et al.* (1996) used a soil phosphorus concentration of 0.03% ± 0.001% and suggested that channel erosion was the major source of phosphorus in Southern Tablelands' catchments. Soil nitrogen concentrations in the Richmond River subcatchment above Casino are 0.17% ± 0.1% and soil phosphorus concentrations are 0.031% ± 0.024% (McGarity and Munns 1955). This simple model does not account for nutrient enrichment caused by selective erosion of fine organic and clay particles that typically have greater nutrient concentrations than bulk soils (e.g. Finlayson and Silburn 1996). This may or may not be important depending on the dominant source of erosion, catchment slope or the size of the rainfall event causing transport of soil particles. In the Jerrabomberra Creek catchment near Canberra, channel incision is the dominant source of sediment yield (Wasson *et al.* 1998). In this instance there may be little opportunity for nutrient enrichment during major transport events when eroded sediments are exported directly by channel flow. In contrast, in catchments with large areas of cultivated land, surface erosion may be the dominant source of sediment being transported (Sharpley and Menzel 1987).

2.2.1.6 Multi-factor empirical modelling approach

A modelling approach was used to derive nitrogen, phosphorus and suspended sediment exports from Queensland coastal catchments (Moss *et al.* 1993). The objectives of the modelling exercise were to (a) determine the relative importance of each particular catchment as a source of nutrients to the coastal environment, (b) evaluate the effects of different land uses, (c) evaluate the relative importance of point sources compared with diffuse sources, (d) evaluate the significant of fertiliser inputs and (e) scope the effects of better land use practices of reduction of sediment and nutrient export. The models employed are expressed in the following equations:

Suspended sediment (t) =

$$L\,[p,g,c;\ km^2] * E\,[p,g,c;\ t\ km^{-2}] * DR\,[p,g,c] * R\,[p,g,c] \qquad (2.5)$$

Nutrient export (t) =

$$L\,[p,g,c;\ km^2] * E\,[p,g,c;\ t\ km^{-2}] * SC\,[p,g,c;\ t\ t^{-1}] * ER\,[p,g,c] * DR\,[p,g,c] * CF\,[p,g,c] * R\,[p,g,c] \qquad (2.6)$$

R [p,g,c] =

$$\text{storm discharge (ML) / catchment area (km}^2\text{)} \qquad (2.7)$$

where L = area of a specified land use
E = erosion rate for a specified land use
DR = delivery ratio for a specified land use
R = runoff correction factor for a specified land use
SC = soil nitrogen or phosphorus content
ER = enrichment ratio (phosphorus only)
CF = dissolved nitrogen or phosphorus compensation factor
p = pristine lands
g = grazing
c = cropping

Moss *et al.* (1993) used erosion rates of 500, 2000 and 5000 t km^{-2} for pristine, grazing and cropping respectively, a delivery ratio of 0.1, an enrichment ratio of 1.5, and a dissolved nitrogen compensation factor of 1.5. They did not consider it necessary to compensate for dissolved phosphorus, making the assumption that the majority of P is transported in particulate forms. About 70% of the nitrogen load in the Richmond River subcatchment is transported as dissolved load, and about 50% of the total phosphorus load is as dissolved inorganic and organic forms. These correspond to CFs of 3 for nitrogen and 2 for phosphorus. Enrichment ratios for total phosphorus range from 1.5 to 8.9 (Sharpley and Menzel 1987). When the delivery ratio is large, the nutrient enrichment ratio is small (Novotny and Chesters 1989; Finlayson and Silburn 1996). A comparison of sediment-bound nutrient transport in the Richmond River subcatchment and soil nutrient concentrations suggest an ER of 1.6 for phosphorus and 0.8 for nitrogen. These calculations assume there is no other source of particulate nitrogen or phosphorus other than catchment soils, a reasonable assumption given that most (> 82%) nutrient and sediment transport occurs during flood periods in the Richmond subcatchment. An ER of less than 1 for nitrogen may reflect an underestimate of catchment soil nitrogen, or a non-diffuse source of particulate nitrogen in river transport.

2.2.1.7 Turbidity as a predictor of nutrient and sediment exports

Turbidity is commonly found to positively correlate with other water quality parameters such as nutrients and suspended sediments. Poor relationships between suspended sediment concentrations and turbidity are commonly found when a low range of suspended sediment concentrations is used, but the relationships improve when a larger range is considered (e.g. Gippel 1989; Eyre *et al.* 1997). Nutrients also show a poor correlation with turbidity in the lower range; this may be due to dissolved nutrient sources (point sources), biological processes, or sediment–water interactions increasing nutrient concentration without any effect on turbidity. Continuous turbidity records collected using automated optical sensors can be calibrated with routinely collected water quality samples to derive a continuous record of sediment concentration (Webb and Walling 1982; Walling *et al.* 1997). The utility of turbidity readings for the calculation of total phosphorus and suspended sediment loads has recently been shown for the Latrobe River catchment (Grayson *et al.* 1996). They stressed that turbidity meters are by no means standard. Although many models may quote NTU units there is much variation between instruments. It was concluded that turbidity readings provided a good predictive approach to measuring loads.

The routinely collected turbidity data provided by the Casino Shire Council and nitrogen, phosphorus, and suspended sediment concentrations collected in the Richmond River test catchment during 1996 were used to develop relationships using least squares regression (Figure 2.6). On days when more than one measurement of suspended sediment concentration was taken, a geometric average was used. High correlations were found between discharge and turbidity, and between suspended sediment and turbidity ($r^2 > 0.85$). The correlations were poorer for nitrogen and phosphorus, although phosphorus showed a much better relationship for turbidity greater than 10 NTU.

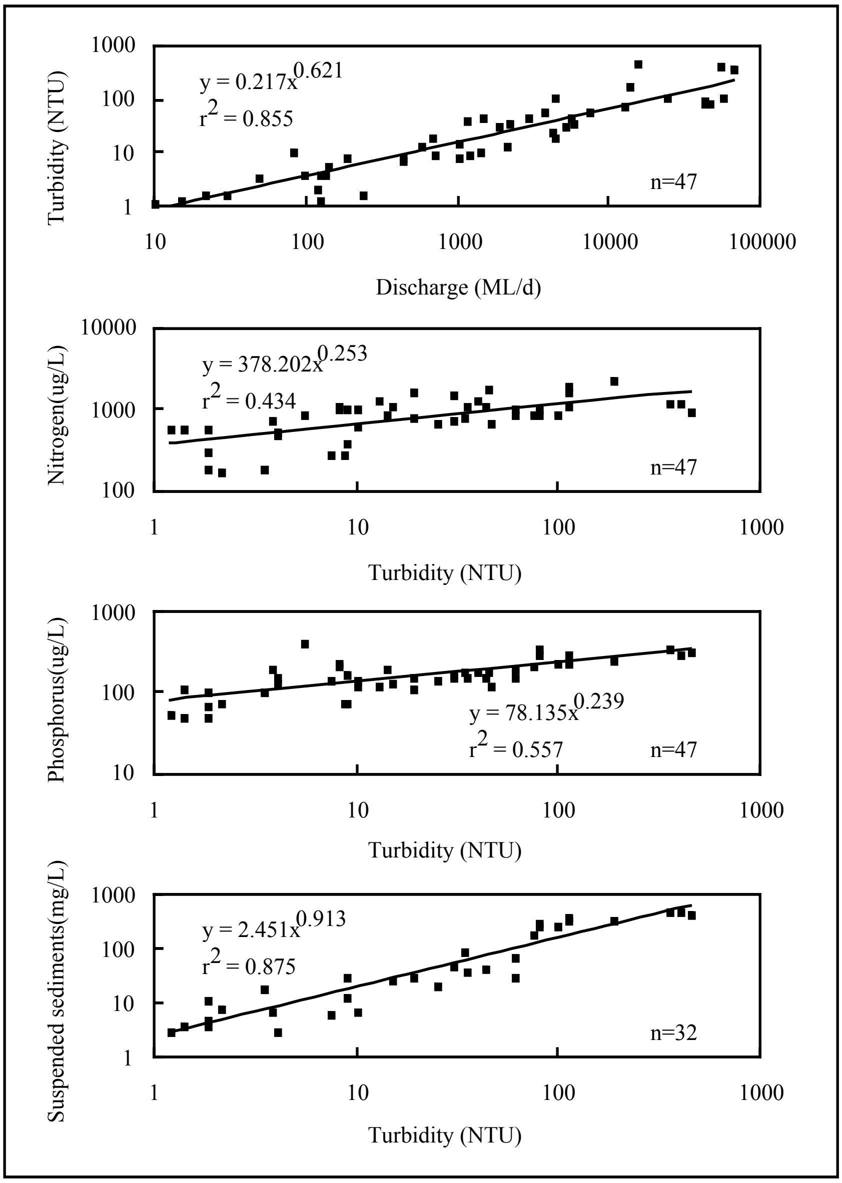

Figure 2.6: The relationships between turbidity, discharge, nitrogen, phosphorus and suspended sediment concentration in the Richmond River test catchment

2.2.1.8 Sediment yield as a function of catchment area and slope

The yield of a catchment per unit area is likely to decrease with increasing area (Ongley 1976; Haith and Shoemaker 1987; Novotny and Chesters 1989; CSIRO 1992; Milliman and Syvitski 1992; Milliman 1995). In a study of Canadian watersheds discharging to Lake Erie, there were also apparent relationships of total nitrogen and total phosphorus yield with basin area for basins between 10 and 10 000 km^2, illustrating the particulate dependence of nutrient transport (Ongley 1976). However, the scatter was large and greater than the inverse trend (especially in the case of nitrogen) and no causal relationships were developed. Milliman and Syvitski (1992) collated sediment yields, basin area, and runoff for 280 world catchments. The catchments were categorised into five groups based on height above sea level: > 3000 m, 1000–3000 m, 500–1000 m, 100–500 m, and < 100 m. For each category, regression relationships were developed with catchment area and catchment runoff (Table 2.6). The relationships with area had correlations between $r^2 = 0.70$ and 0.82, but the relationships for runoff were poorer (r^2 = 0.36 to 0.66). The regression equations for the 100–500 m category were used to estimate sediment yield for the Richmond River test catchment.

Table 2.6: Regression relationships between catchment area, runoff, and sediment yield for each topographic category (Milliman and Syvitski 1992)

Height (m a.s.l)	Description	Regression equations (Area)	Regression equations (Rainfall)
		Load (1×10^6 t yr^{-1})	Yield (t km^{-2}yr^{-1})
> 3000	High mountain	280 $A^{0.46}$	0.5 $R^{1.16}$
1000–3000	Mountain	170 $A^{0.52}$	20 $R^{0.65}$
500–1000	Upland	12 $A^{0.42}$	0.002 $R^{1.74}$
100–500	Lowland	8 $A^{0.66}$	0.002 $R^{1.67}$
< 100	Coastal plain	1 $A^{0.64}$	0.001 $R^{1.57}$

A catchment area
R rainfall

2.2.2 Results

Estimates of nutrient loads for the test catchment using each empirical method are compared with the ‘real’ loads calculated using linear interpolation (Table 2.7). The simple empirical regression models both over- and underestimated nutrient exports in the test catchment. Estimates made using population density (Table 2.7; E1) underestimated both nitrogen and phosphorus export by 0.46 and 0.08 ×. The fertiliser loading regression overestimated nitrogen (1.41 ×) and underestimated phosphorus exports (0.59 ×). Nutrient and sediment interrelationships (Table 2.7; E3) both overestimated by up to 3.2 × and underestimated by up to 0.18 ×. Land export relationships for nitrogen underestimated nitrogen export using average rates given for tropical Australian catchments (Young *et al.* 1996). When the upper rates were used (Table 2.5), nitrogen export was slightly overestimated (1.23 ×) and phosphorus and suspended sediments were underestimated. Sediment yield and catchment soil nutrient content (Table 2.7; E5) underestimated both nitrogen and phosphorus export. It is likely that particulate matter being exported is enriched in phosphorus compared to catchment soils. The other reason for an underestimate by this method is that there are additional sources of nutrients in the catchment such as human sewage and animal effluent which are unrelated to soil nutrient concentrations.

Table 2.7: A comparison of estimated nutrient and suspended sediment exports in the test catchment generated using empirical models and the real exports calculated by linear interpolation

(The factor difference is calculated as the load of the method in question divided by the load estimated by linear interpolation.)

Model	Description	Total N (t)	Factor difference	Total P (t)	Factor difference	SS* (10^3 t)	Factor difference
Real	Linear interpolation	606	–	142	–	141	–
E1	Population density and riverine nutrient export	278	0.46	12	0.08	–	–
E2	Fertiliser loading and riverine nutrient export	855	1.41	84	0.59	–	–
E3	Nutrient and sediment export interrelationships	P-N 1083 SS-N 1942	1.79 3.20	N-P 75 SS-P 343	0.53 2.42	N-SS 26 P-SS 40	0.18 0.28
E4	Land use export relationships	747**	1.23	87**	0.61	113	0.80
E5	Sediment yield and soil nutrient content	381***	0.63	77***	0.54	–	–
E6	Moss *et al.* (1993) 'Model 2'	602	0.99	146	1.03	147	1.04
E7	Turbidity and nutrient and sediment relationships	707	1.17	137	0.96	127	0.90
E8a	Sediment yield and runoff	–	–	–	–	62	0.44
E8b	Sediment yield and catchment area	–	–	–	–	123	0.87

* suspended sediment
** calculated using upper limits of nutrient and sediment generation rates (Table 2.5)
*** calculated using upper limits of soil nutrient concentration in the Richmond catchment

The model of Moss *et al.* (1993) (here denoted E6) takes into account both enrichment and nutrient sources other than soils, using an enrichment ratio for phosphorus and a dissolved nutrient compensation factor for nitrogen and phosphorus. Loads estimated using this method agree with the 'real' loads within 4%. Nutrient and suspended sediment exports generated using routinely collected turbidity measurements also show a reasonably close comparison to the exports calculated using linear interpolation. The nitrogen export was positively biased by 17%, whereas phosphorus and suspended sediment were negatively biased by 4% and 10% respectively. The sediment yield verses runoff model (Table 2.7; E8a) underestimated sediment yield in the Richmond test catchment by 0.44 ×, whereas the relationship between catchment area and sediment load showed a negative bias of 13%.

Of the range of empirical methods tested, the model demonstrated by Moss *et al.* (1993) appears to give the best estimate of nutrient and sediment exports from the test catchment (within 4% of the loads calculated using linear interpolation). Regression relationships between turbidity and pollutant loads also appear robust.

2.3 Discussion

2.3.1 Performance of the tested methods

The previous sections have described a range of methods (mathematical equations and empirical models) that have been described in scientific literature. Some of these methods were compared to 'real' nutrient and sediment loads (calculated by linear interpolation) using a subcatchment of the Richmond River catchment, northern NSW, for which there was a data set available (Figure 2.7). Nitrogen load in the test catchment was estimated by 18 methods, 11 of which gave estimates within 25% of the load calculated by linear interpolation (Figure 2.7; LI). Phosphorus load in the test catchment was also estimated using 18 methods, 10 of which gave estimates within 25% of the load calculated by linear interpolation. Suspended sediment loads were estimated using 17 methods, 9 of which gave an estimate within 25% of the real loads calculated by linear interpolation. The poorest estimates for pollutant loads were generated using the two averaging methods (1, 2) which used time factors to scale loads or concentrations up to the annual time scale. These scale methods are not suitable for use with the stratified data set that exists for the Richmond test catchment because sampling was biased towards high flow. Estimates were improved using flow as a scaling factor (Methods 1a, 2a, equivalent to Methods 7, 15).

The empirical methods using relationships between nutrient export and population (E1) or fertiliser loading (E2) were generally poor predictors of nutrient load for the test catchment. These methods are probably more suited to regional-scale studies rather than for catchment-scale estimates where highly localised conditions prevail. They also show the likely effects that changes in catchment land use and management could have over time as population intensity and the demand for food production increases.

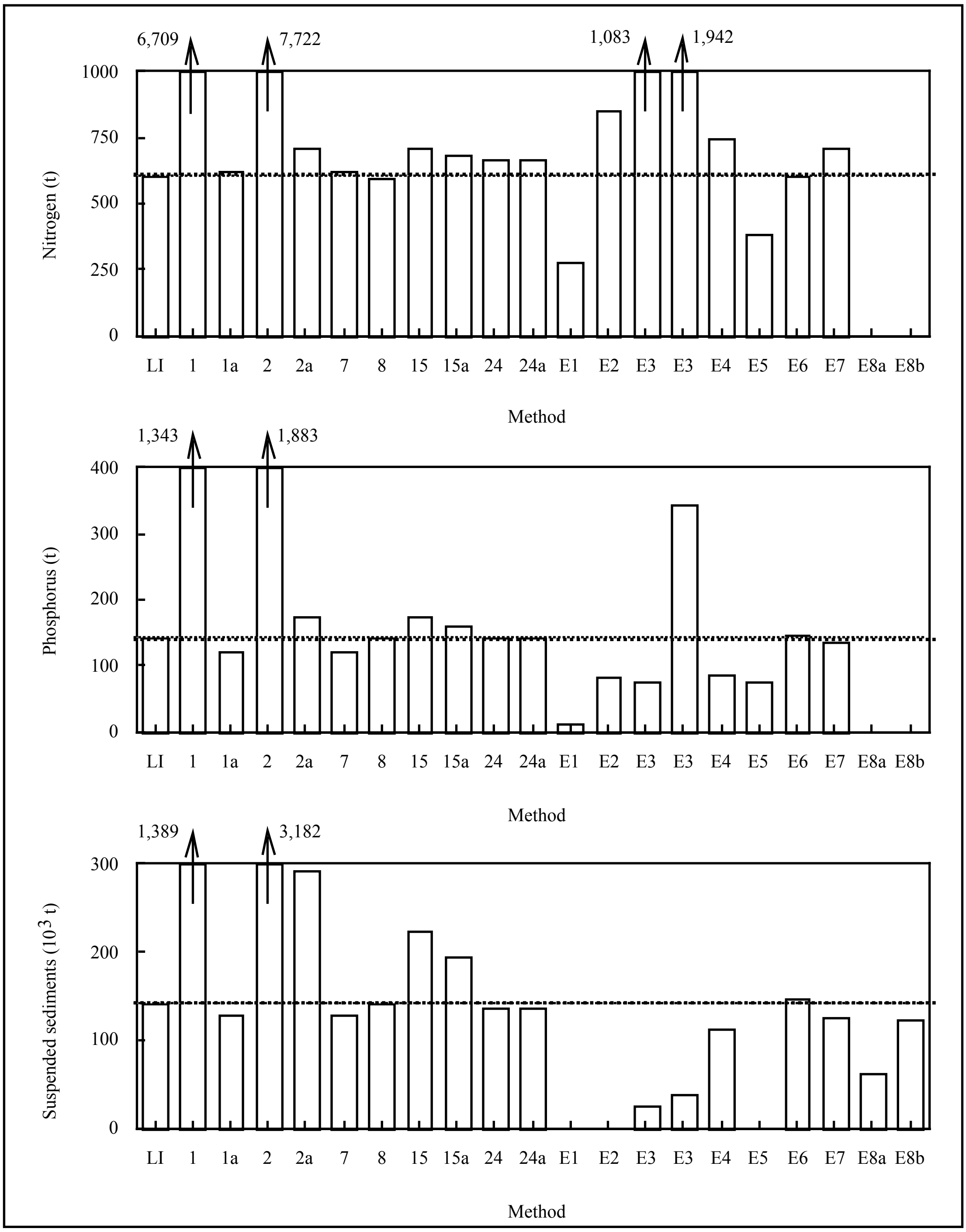

Figure 2.7: A comparison of nutrient and sediment loads generated using a range of methods for the Richmond River test catchment

(The dashed line is the load calculated using linear interpolation)

2.3.2 Choosing the right method

There are a wide variety of methods available (mathematical, empirical, or computer modelling) for calculating nutrient and sediment loads of catchments. No method will give an accurate and unbiased assessment for all situations and therefore it is important to choose the right method. The appropriateness of each method is dependent on data availability, the hydrological characteristics of the catchment, catchment area, and the complexity of land use, soil types, and interacting morphological characteristics such as topography. Further constraints include financial and technological issues, however, these will not be considered here.

Data availability can be classified into four groups:
1. stratified or flow-weighted data
2. routine data with spot sampling during flood flow
3. routine data
4. no data.

2.3.2.1 Stratified or flow-weighted data

Stratified or flow-weighted data is the best class of data available and characterises concentration over the entire range of flow conditions (e.g. the Richmond River test catchment). This sort of data is usually collected only in a flow-gauged catchment. Flow-weighted data is best collected in small or quick response catchments using automated techniques such as optical turbidity sensors (calibrated using routine or spot sampling) or automated samplers that store and preserve water samples for later laboratory analysis. In larger catchments, or in catchments that have only a few storm events during the annual cycle such as the Richmond River catchment, water sampling can be done manually taking 2–6 samples per day over the storm flow period, which may last for 3–15 days. The best method for calculating nutrient or sediment loads with data that characterises the range of flow conditions is linear interpolation, but Methods 8, 15a, 24, and 24a (Table 2.4) should also give an accurate measure of annual load. The problem with manually collected data in this category is that it usually covers a small time period of 1 or 2 years when there are initiatives for collection. The period of data collection may not always coincide with 'typical' catchment conditions, for example data collected during a drought year. The utility may be extended when used to calibrate predictive models (e.g. Moss *et al.* 1993).

2.3.2.2 Routine data with spot sampling

The next best category is routine data (e.g. collected daily weekly, fortnightly, monthly, seasonally) combined with spot sampling during high flow. This sort of data is often collected by government bodies and water supply authorities to fulfil State of the Environment objectives and regional water quality assessments (e.g. *The Northern Rivers—A Water Quality Assessment*, conducted and published by the NSW EPA 1996). Calculation of annual pollutant loads using routine data with spot sampling of high flow will often depend on the availability of discharge data. In the event that discharge data is available or can be calculated using a gauging station close by, nutrient loads can be calculated using methods 8, 15, and 24. Methods 1a and 2a may also give a reliable estimate. Given the relative ease of these methods, it would probably be appropriate to use more than one method and compare the results.

2.3.2.3 Routine data

Routine collection is the most common method for assessing water quality. Such data is commonly collected by government departments and local councils for resource assessment and State of the Environment reporting. This sort of data is usually collected on a spatial or regional basis covering many locations within the jurisdiction of the agency or department concerned. Data are often compared to water quality guidelines (ANZECC 1992).

Turbidity data collected on a daily basis constitutes a special case when routine data collection may produce very reliable load estimates (e.g. the Richmond River at Casino). Turbidity data is collected routinely in many catchments of NSW in which water supply is drawn from the river adjacent to a town. It is likely that data may exist or could be collected using spot sampling over the full range of seasonal conditions (including flood sampling) to calibrate routinely collected turbidity data sets. A set of likely data sources was collected from the Water Cycle Planning Management Section, NSW Department of Land and Water Conservation, Parramatta. Although it seems likely that any local authority drawing river water for town water supply would need to measure turbidity, no attempt has been made to verify the availability of data from each location.

Sometimes routine data is collected from a single site routinely on a time frame from days to weeks, and if discharge data is available or reliable estimates of discharge can be made, an estimate of pollutant load may be calculated. In this situation, an assessment of the representativeness of the concentration data over the range of river discharges would be needed. If several of the routinely collected concentration data happened to coincide with flood discharge it would be possible to use Methods 1a, 2a, 8, 15, and 24 to estimate pollutant loads. Again it would be best to use several methods to help assess the reliability of the estimates.

In the event that none of the routinely collected samples coincided with storm discharge, there are several situations when reliable estimates may still be made:

1. In a regulated river system where discharge variability was low due to the operation procedures of an upstream dam.
2. When there was no flood flow for the period in question.

In the event that none of the previous criteria were fulfilled, load estimates would be tenuous. In this case it may be possible to use the data to help calibrate a computer model.

2.3.2.4 No data

When there is no data available (most situations), empirical models and computer models can be used to estimate loads. For example, the computer model AQUALM has commonly been used in Australia for estimating non-point source loads. AQUALM is an example of a model that generates runoff and pollutant exports using daily time steps, then routes them through a stream network. Pollutant generation relies on area and distribution of broad land use categories and therefore the model does not take into account nutrient and sediment generation from streambank erosion, gullies, or other potential catchment specific sources (Wasson *et al.* 1996).

A comparison of nutrient and sediment loads generated by computer models and loads calculated using field data reveals large discrepancies for a range of catchments on the east coast of Australia (Table 2.8). For catchments in south-eastern Queensland, AQUALM both underestimated and overestimated nutrient loads by up to a factor of 10.8 ×. In the case of sediment loads, AQUALM estimates can be in error by up to 9 × compared to field measured loads. Typically AQUALM was found to overestimate in near-natural catchment and underestimate in disturbed catchments (Wasson *et al.* 1996). In contrast, Eyre and McKee (1998) found that AQUALM overestimated nutrient loads in south-eastern Queensland in all cases but two (Table 2.8). Wasson *et al.* (1996) suggested that the reason for the poor predictive ability of models when applied to Australian catchments was the lack of consideration for channel and gully erosion (which are important nutrient and sediment sources in many Australian systems) and the large overlap of phosphorus generation from native pasture, improved pasture, and dryland cropping, three common Australian farming practices.

The comparisons between real loads and CMSS were reasonable in the Richmond River catchment (Bungawalbin, Richmond, Wilsons, and Coastal subcatchments), but in this case a knowledge of the real loads was used to ‘calibrate’ the CMSS output. During the calibration process both the areas of each land use and the nutrient generation rate for each land use were adjusted. The CMSS handbook contains a number of comparisons between measured and CMSS estimated loads for Onkaparinga, SA, Hawkesbury–Nepean, NSW, Peel–Harvey, WA, and Wyong, NSW. In these examples CMSS both over- and underestimated loads of nitrogen and phosphorus by up to 43 times. However, loads for large catchments were estimated more accurately.

Another reason for poor correlation between modelled and measured nutrient and sediment loads may derive from inaccurate measurement and interpretation of areas in each land use class. Both AQUALM and CMSS rely on nutrient generation rates applied to a particular land use category. The area of each land use is usually interpreted from areal photographs and / or satellite imagery with little or no ground truth. There can be overlap between land use classes. For instance, low-intensity grazing may occur under open-canopy forest; closed-canopy rainforest may be either logged or undisturbed; rural grazing may be in part rural residential. The majority of land use data available has not been collected with the objectives of nutrient and sediment generation rates in mind, and land use patterns are dynamic. Further problems can occur by grouping land use in broad categories. For instance, urban areas may have concrete or grass swale drainage systems and may be sewered or unsewered; dairy farming on clay soils that adsorb phosphorus may release less phosphorus and nitrogen than low-intensity grazing on sandy soils with a high leaching potential. Some grazing or cropping areas may have intact riparian vegetation on streams whereas other areas will be closely connected to the stream with farming on the riverbanks. For example, a greater nutrient generation rate was chosen for areas < 50 m from the riverbank than for areas of similar land use > 50 m from the river when modelling nutrient loads in a catchment in southern England (Heathwaite *et al.* 1996). Choosing the right generation rate is therefore always problematic.

There are many problems associated with modelling nutrient and sediment loads from catchments with no data available, evidenced by the comparisons made in Table 2.8. It would seem difficult to justify the cost of modelling when order-of-magnitude estimates or better can be made using empirical methods E1, E2, E3, E8b (Table 2.7). The empirical methods also allow relative ranking of catchments or subcatchments so that areas at risk of higher nutrient and sediment loads are easily determined. In catchments with no data collection, the empirical model E6 (Moss *et al.* 1993) shows the best potential for nutrient and sediment load estimation in the Richmond River test catchment. There are a number of existing data sets held at Southern Cross University which could be used to develop a NSW-specific empirical model (similar to Model E6) that is capable of predicting nutrient and sediment export. There are 11 data sets listed in the CMSS handbook for the Hawkesbury–Nepean and one from the Wyong catchment, NSW. There are also published data from Lake Burley Griffin (Cullen 1978), the Hunter Valley, and the Southern Tablelands (Wasson 1998). The advantages of models like E6 is that they are fast, cost-effective and reasonably accurate, require little data, and allow catchments to be ranked in terms of risk of nutrient and sediment export.

Table 2.8: Comparisons of modelled nutrient and sediment loads with loads generated using field data collection and discharge

(Factor difference is the modelled estimate divided by the sampling estimate.
N—nitrogen; P—phosphorus; SS—suspended sediment)

Model	Catchment, year	Parameter	Model estimate (t)	Sampling estimate (t)	Factor difference	Reference
AQUALM	Logan, 1996	N	1400	777	1.8	Eyre and McKee (1998)
		P	140	203	0.7	
	Logan, 1997/98	N	550	51	10.8	
		P	79	20	4.0	
	Brisbane, 1996	N	4500	2220	2.0	
		P	620	342	1.8	
	Caboolture, 1996	N	190	348	0.5	
		P	29	24	1.2	
AQUALM	L.Goodradigbee	SS	1990	260	7.7	Wasson *et al.* (1996)
	L. Yass	SS	9250	24000	0.4	
	Woodstock	SS	260	1110	0.2	
	Sturt	SS	120	260	0.5	
	L. Queanbeyan	SS	330	2290	0.1	
	Uriarra	SS	930	5900	0.2	
	Williamsdale	SS	1130	2660	0.4	
	Buchan	SS	400	3710	0.1	
	L. Numeralla	SS	1380	980	1.4	
	Adaminaby	SS	800	410	2.0	
	Yaouk	SS	40	170	0.2	
	Bungawalbin	P	49	26	1.9	McKee (unpubl.); Anon. (1998)
	Richmond	P	187	211	0.9	
	Wilsons	P	130	172	0.7	
	Coastal	P	118	89	1.3	
CMSS	Brunswick, N. NSW	N	20	25	0.8	Pont and Eyre (1997)
		P	3.4	1.6	2.1	

3. EROSION AND SEDIMENT/NUTRIENT TRANSPORT MODELLING

3.1 Types of models

A large number of sediment transport / water quality models exist. These models differ both by the types of water quality issues they address (e.g. phosphorus, suspended sediment), and by the level of physical processes simulated by the model. In general, models fall into three main categories, depending on the physical processes simulated by the model and the data dependence of the model.

1. Empirical/metric models—these models are based on simple empirical relationships between sediment and nutrient generation and catchment characteristics such as climate and land use, with little or no attempt being made to describe the physical processes occurring within the catchment system. (e.g. CMSS).
2. Conceptual models—these models are based on a generalised concept of the catchment as a configuration of internal storages through which flows and pollutants pass. The level of sophistication in the description of physical processes, both hydrological and hydrochemical, varies between models (e.g. LASCAM, HSPF, IHACRES-STARS).
3. Physics-based models—these models are based on the solution of so-called fundamental equations of sediment transport and catchment response, and usually assume detailed representations of the processes driving sediment and nutrient generation and transport (e.g. ANSWERS, WEPP).

All three model types have inherent limitations and advantages in their application. The best model for a problem will depend on factors such as the scale of the problem, the availability of data, the intended use of the model, and the user's skills and resources. The distinction between models is not sharp. For example, some conceptual models may have empirical components, whilst other models may be much more physically based while still falling into the category of conceptual models. For this reason there may be disagreements in the literature as to which category a particular model belongs to. Models may also be described as hybrids between two of these classes, e.g. hybrid metric/conceptual when the model consists of significant empirical and conceptual components. Also, the classification categories and definitions are not universally agreed upon by the modelling community, so different authors may use different classifications of models.

Erosion, sediment and nutrient transport models generally consist of both hydrological and nutrient/sediment transport components. One major difference between specific models is the complexity of treatment of rainfall-runoff in the sediment and nutrient generation process. Models such as CMSS do not attempt to model the hydrology of the catchment system, whereas other models, such as WEPP, include a rainfall-runoff model within their structure.

Different types of models are also subject to different sources of uncertainty and error. Errors produced in model outputs may be either systematic or random. A systematic error is non-random and introduces bias into model outputs. For example, a model suffering from the presence of systematic errors will often consistently underpredict or overpredict outcomes. Errors in model output can arise from a number of sources, including misspecification of physical processes within the model structure and inaccurate measurement of model inputs such as sediment and nutrient concentrations and flow discharges, or from uncertainties involved with sediment and nutrient load estimations used for model calibration. Some sources of error will be a problem with any type of model selected, whereas other sources may be eliminated or reduced by using a different model type.

3.1.1 Empirical/metric models

Empirical models are generally the simplest of all three model types. This means that the computational and data requirements for such models are usually smaller than for conceptual and physics-based models, empirical models generally being capable of being supported by coarse measurements. Jakeman *et al.* (1997) state that 'the feature of this class of models is their high level of spatial and temporal aggregation and their incorporation of a small number of causal variables'. Many empirical models are based on the analysis of catchment data using statistical techniques, and as such are ideal tools for the analysis of data within catchments. Such models are particularly useful as a first step in identifying sources of sediment and nutrient generation.

Most empirical models do not attempt to represent the physical processes involved in sediment generation. For this reason many empirical models tend to be catchment-specific, that is, they apply only to the catchment for which they have been developed, and often under the specific land use conditions existing within the catchment at that time. This means that the ability of empirical models to predict the effects of changes in catchment characteristics, such as land use, on water quality and sediment yields can be limited. Empirical models also tend not to be event-responsive (or responsive to antecedent conditions), ignoring the processes of rainfall-runoff in the catchment being modelled.

Empirical models are often criticised for employing unrealistic assumptions about the physics of the catchment system, ignoring the heterogeneity of catchment inputs and characteristics, such as rainfall and soil types, and the inherent nonlinearities in the response of the catchment system. Such models are also generally based on the assumption that underlying conditions remain unchanged for the duration of the study period.

Walton and Hunter (1996) mention two empirical models, CMSS and AEAM. They state that these models rely on estimation of parameters through either local knowledge or expert knowledge or from previous model application. They suggest that these models are intended only as initial planning tools and state that 'they do not provide highly accurate prediction of water quality or quantity'.

3.1.2 Conceptual models

Conceptual models are typically based on the representation of the catchment as a configuration of internal storages and pathways. Conceptual models usually incorporate the (catchment-scale) underlying physical mechanisms of sediment and runoff generation within their structure, representing flow paths within the catchment as a series of storages, each requiring some characterisation of its dynamic behaviour. Conceptual models are not generally spatially distributed, although this is not necessarily the case (Jakeman *et al.* 1997). Rather, conceptual models tend to lump representative processes over the scale at which outputs are simulated. Parameter values for conceptual models have typically been obtained through calibration against observed data such as stream discharge and concentration measurements.

Due to the requirement that parameter values be determined through calibration against observed data, conceptual models tend to suffer from problems associated with the identifiability of their parameter values. Most calibration techniques used for conceptual models of medium complexity (say more than half a dozen parameters) are capable of finding only local optima at best. Often calibration of parameters in a conceptual model identifies only a set of sufficiently accurate parameter values which reproduce observed behaviour in some sense, not necessarily a globally optimal set. This means that there are many possible 'best' parameter sets available. Spear (1995) notes this problem in large simulation models stating that 'there is not a single point in the parameter space associated with good simulations, indeed there generally is not even a well-defined region in the sense of a compact region interior to the prior parameter space'. Thus the likelihood of identifying a unique 'best' parameter set, in terms of goodness of fit, is very small. Increasing model complexity tends to decrease *a priori* model identifiability (Kleissen *et al*. 1990). This means that in general, simpler conceptual models have fewer problems with model identification than more complex models. Thus problems with model identification can be minimised through limiting the number of parameters to be estimated through calibration and possibly identifying additional parameters using *a priori* knowledge of the system. However this reduction in problems associated with identifiability through simplification of models *may* come at the expense of goodness of fit to calibration data. More complex models are more likely to provide a better fit to calibration data, although this does not necessarily extend to providing better predictions of future behaviour, as complex models run the risk of overfitting calibration data (see the end of section 3.1.3).

The lack of uniqueness in parameter values for conceptual models means that the parameters in such models have limited physical interpretability. However, this problem can also be associated with empirical and physics-based models. Physics-based models in particular are often over-parameterised whereas empirical models tend to be naturally much simpler in their level of parameterisation.

3.1.3 Physics-based models

Physics-based models are grounded on the solution of fundamental physical equations describing streamflow and sediment and nutrient generation within the catchment. Standard equations used in such physics-based models are the equations of conservation of mass and momentum for flow and the equation of conservation of mass for sediment (e.g. Bennett 1974).

In theory, the parameters used in physics-based models are measurable within the catchment and so are ‘known’. However, in practice, the large number of parameters involved and the heterogeneity of important characteristics within the catchment means that these parameters must typically be calibrated against observed data. This creates additional uncertainty in parameter values. Also, even in situations where parameters can be ‘measured’ within the catchment, errors in the measurement of important characteristics will create additional uncertainty as to the veracity of model outcomes. Where parameters cannot be measured within the catchment they must be determined through calibration against observed data. Given the large number (possibly hundreds) of parameter values needed to be estimated using such a process, problems with the lack of identifiability of model parameters and non-uniqueness of ‘best fit’ solutions can be expected. There is likely to be a large number of parameter values for which the model gives an adequate fit. Thus the physical interpretability of model parameters is questionable. In the case of large simulation models, where many possible ‘best’ parameter sets are available, there are ‘clear limitations on how one might interpret the technical or scientific significance of any particular set of parameters that lead to a good fit’ (Spear 1995). In the case of physics-based models this means that the necessity to calibrate some or all parameter values will undermine the physical interpretability of the entire parameter set.

An additional problem with estimating model parameters in physics-based models is the necessity to lump together spatially distributed variables into data at a single point. Lane *et al.* (1995) state that ‘model parameters derived in this manner represent nothing more than fitted coefficients distorted beyond any physical significance’. In general the equations governing the processes in physics-based models are derived for small-scale models under very specific physical conditions. However, in physically based models these equations are used at much greater scales, and under different physical conditions. The equations are generally derived for use with continuous spatial and temporal data, but the data used in these models is often point-source data taken to represent an entire grid cell within the catchment. The derivation of mathematical expressions describing individual processes in physics-based models is subject to numerous assumptions that may not be relevant in many real-world situations (Dunin 1975). The viability of lumping up small-scale physics to the scale of the spatial grid used in many physics-based models is also questionable (Beven 1989). Specifically there is a lack of theoretical justification for assuming that equations apply equally well at the grid scale, at which they are representing the lumped aggregate of heterogeneous subgrid processes (Beven 1989).

Physics-based models also tend to have greater data and computational requirements than other model types. Parameter values must be measured both spatially and temporally within the catchment. The use of such models has been limited by the lack of observed physical and biological data within catchments, and by the larger computing costs involved in their use.

The tradeoff between model complexity and accuracy is not simply that increased model complexity increases model accuracy. Simpler catchment models perform equally well or at least are not substantially outperformed by more complex models (Loague and Freeze 1985). Jakeman and Hornberger (1993) confirmed this result for different levels of complexity in conceptual models.

3.2 Specific erosion and sediment/nutrient models

Many different erosion and sediment/nutrient transport models are currently available. These models differ in complexity, the catchment processes modelled and the assumptions on which they are based. This section provides an outline of a number of currently available models, including information on their cost, availability and hardware requirements.

3.2.1 AEAM

The Adaptive Environmental Assessment and Measurement program (AEAM) is a process for the development and exploration of management options for complex systems. One of the main outcomes of the AEAM process is the development of a model based on expert knowledge of the system. The complexity of the model developed depends on the relationships within the system considered to be necessary by the expert groups consulted. Grayson *et al.* (1994) identifies AEAM as being 'a philosophical and methodological framework designed to deal with the uncertainties inherent in environmental changes'. The AEAM approach relies on expert knowledge along with historical variability and patterns of change to characterise the system. The initial simulation model developed from this characterisation is used to design management programs which measure responses to management actions, which are used to refine the initial model. In this way, the model is an adaptive approach to catchment management.

There is not a set model structure for AEAM. Instead, the program can be thought of as a guideline for catchment management groups to approach their water quality problem. The basic design of the AEAM program is in two parts. The 'shell' handles the input/output and provides the structure to manage the spatial and temporal data, while the 'dynamic simulation' performs the numerical simulation of the system (Grayson *et al.* 1994). The shell tends to be generic between AEAM models, while the dynamic simulation is developed for the specific application.

AEAM, although widely used elsewhere, has not been used to a large extent in Australia (Grayson *et al.* 1993; Grayson *et al.* 1994). AEAM models of catchment behaviour are generally simple balance-type representations based on rainfall and evaporation input data. The models developed do not normally attempt to quantify the processes involved in water quality and are not formulated as predictive or forecasting tools. Instead, they are a more trial-and-error approach to catchment management, generally relying on empirical and simple conceptual models. Like CMSS, these models rely on calibration of parameter estimates and are intended only as planning tools (Walton and Hunter 1996).

The model has been used for integrated catchment management on the Latrobe and Goulburn River catchments (Grayson *et al.* 1994 and Grayson and Doolan 1995 respectively) and for the improved integration of planned riparian-zone research in the North Johnstone catchment in Queensland (Argent and Wilson 1996).

Examples of model users:
LWRRDC; Centre for Environmental Applied Hydrology at the University of Melbourne

Hardware Requirements:
Run under QuickBASIC or VisualBASIC on PC

Availability/Cost:
Models are generally adapted from a model used in a similar AEAM application elsewhere. LWRRDC and the Centre for Environmental Applied Hydrology at the University of Melbourne have prepared Occasional Paper 01/95, 'Adaptive Environmental Assessment and Measurement [AEAM] and Integrated Catchment Management' ($28), providing more detailed information and two Victorian applications for the Latrobe River and Goulburn and Broken Rivers catchments on PC-format floppy discs.

For further information contact:
Dr Rodger Grayson
Phone: (03) 9344 7305
Email: rodger@civag.unimelb.edu.au

3.2.2 AGNPS

The Agricultural Non-Point Source model (AGNPS) is an event-based, non-point source pollution model developed by the US Department of Agriculture's Agricultural Research Service (USDA-ARS) in cooperation with the Minnesota Pollution Control Agency and the US Soil Conservation Service (SCS). The model was developed to predict and analyse runoff water quality from rural catchments ranging from a few to over 20 000 ha.

AGNPS uses a grid cell representation of the catchment, with cell resolution ranging from 0.4 to 16 hectares. Runoff and transport of sediment, nutrient and chemical oxygen demand are simulated for each grid cell, with potential pollutants being routed through cells to the catchment outlet.

Runoff within the catchment is simulated using the SCS curve number method, an empirical rainfall-runoff modelling technique. This method deals with baseflow separately and combines channel runoff, surface runoff and subsurface flow into 'direct' runoff. The rainfall-runoff equation is

$$R_a = \frac{(Q_m - I_a)^2}{(Q_m - I_a) + S_m} \tag{3.1}$$

where R_a is the annual rainfall, Q_m is the potential maximum runoff, I_a is the initial abstraction (often assumed to be $0.2S_m$) and S_m is the potential maximum retention. The value of S_m is related to a curve number CN by the relationship:

$$CN = \frac{1000}{S_m + 10} \tag{3.2}$$

Erosion and sediment transport are modelled using a modified version of the Universal Soil Loss Equation (USLE), which is discussed in sections 3.2.14 and 3.2.17. Two different versions of the AGNPS model have been developed by the USDA-ARS. One uses the USLE, the other the RUSLE (AGNPS98). A version is currently being developed using the USLE-M (Kinnell and Risse 1998, Kinnell, 1998a and 1998b). Soil loss is calculated within AGNPS for each cell in the catchment. The chemical transport component of AGNPS models the transport of nitrogen, phosphorus and chemical oxygen throughout the catchment, using relationships adapted from the CREAMS model and from a feedlot evaluation model. The nitrogen cycle is considered explicitly in AGNPS (Ball and Trudgill 1995). The model treats nutrients and chemical oxygen delivered from feedlots as point sources, and routes them with contributions from non-point sources. Other point source inputs of pollutants and water are modelled by inputting incoming flow and nutrients to the cells in which they occur.

Input data for the AGNPS model includes variables describing catchment morphology, land use variables and precipitation data, generally input for each cell in the catchment grid. The model outputs total volumes associated with runoff, sediment yield and chemical output in a number of different forms, including graphical and numerical representations.

AGNPS was developed and tested on catchments in the USDA but has been applied in a number of different studies on catchments in Australia (Rosewell 1995) and around the world.

AGNPS is generally more accurate in its predictions and analysis of sediment yield than models such as CMSS, but the greater data requirements and computational complexity of AGNPS must be weighed against this improvement in accuracy.

Examples of Model Users:
Department of Conservation and Land Management - Gunnedah Research Station

Hardware Requirements:
PC or Unix/Solaris 2.5×

Availability/Cost:
Freely available from USDA-ARS via the Internet at
http://www.coe.odu.edu/cee/model/agnps.html.

For further information contact:
Mr. Richard Beecham
Phone: (02) 9895 7169
Email: fchs@engineering.unimelb.edu.au

3.2.3 ANSWERS

From the mid 1980s, advances in sediment and nutrient transport modelling included the development of a grid or cellular approach, dividing the landscape into cells which were modelled individually and totalled for the catchment. This approach subsequently provided a common basis for the structure of process-based hydrologic and water quality models (Moore and Gallant 1991). The pioneering model was the Areal Non-Point Source Watershed Environment Response Simulation (ANSWERS) program, a precursor to GIS (Zhang *et al.* 1995). The primary outputs of model simulation are runoff and erosion (Fisher *et al.* 1997), although the model has been extended to include nutrients (Moore and Gallant 1991).

The model uses four main categories of landform parameters: soil, land uses, elevation-based slope and aspect, and channel descriptions in addition to the storm event details (Fisher *et al.* 1997). Within these broad categories many parameters are required. For example, for each soil type the following eight variables are required: total porosity, field capacity, steady state infiltration, the difference between steady state and maximum infiltration, the rate of decrease in infiltration with an increase in soil moisture, infiltration control zone depth, antecedent soil moisture, and erodibility.

The erosion module in ANSWERS is governed by the continuity equation

$$\frac{dM}{dx} = D_{cf} + q_1 \tag{3.4}$$

where D_{cf} is the net detachment or deposition rate and q_1 is the lateral inflow of sediment load to the channel. Detachment of soil particles by raindrop impact is calculated using the relationship

$$R_d = 0.027\, C\, K\, A_i\, R_r^2 \tag{3.5}$$

where R_d is the rainfall impact detachment rate, C is the cropping and management factor of the USLE, K is the soil erodibility factor, A_i is the area increment and R_r is the rainfall intensity.

ANSWERS uses a form of the Yalins' (1963) bedload transport equation to predict the transport of cohesionless grains over a movable bed for steady uniform flow of a viscous fluid (Loch *et al.* 1989b). The extended version of ANSWERS is capable of simulating the transport of individual particle size classes (Rose and Ghadiri 1991).

The model is both temporally and spatially distributed, providing an advantage over less complex models like USLE (Zhang *et al.* 1995). Given the large data requirements for the model simulation, the incorporation of GIS is of increasing importance in large-scale sediment transport or water quality prediction. The effects of rainfall intensity and spatial variation in soil infiltration capacity, surface conditions and topography are explicitly represented by ANSWERS (Connolly *et al.* 1997).

The applicability of ANSWERS is limited in most catchments by the large input data requirements, both spatial and temporal, of the model. Given the lack of such data in most catchments, parameters may need to be calibrated, raising problems with model identifiability and the physical interpretability of model parameters. There are also other potential problems with the program. Fisher *et al.* (1997) concluded from a spatial sensitivity analysis of the model that many outputs were insensitive to changes in the spatial distribution of input variables to the model. The authors proposed three possible explanations: lack of variability of important parameters within the study catchment; key model components unaccounted for; or variables not subjected to spatial mixing in any run swamping the effect of mixing. These findings indicate the possible shortcomings of the model in effectively modelling the processes addressed by the model (Fisher *et al.* 1997). Additionally, ANSWERS considers erodibility to be a relatively time-constant parameter, contrary to the large variations in this parameter that have been recorded (Govers and Loch 1993). This assumption is likely to limit the effectiveness of the model in predicting runoff and soil erosion.

The model has been extensively used by the Queensland Department of Primary Industries for the prediction of runoff from rainfall simulators (Silburn and Connolly 1995; Connolly and Silburn 1995; Connolly *et al.* 1997) and for validation and calibration of a predictive infiltration model and peak discharge estimation models (Silburn *et al.* 1990; Titmarsh *et al.* 1990). The model used was modified to include the Green and Ampt infiltration equation. The work by the Queensland Department of Primary Industries showed the ability of physics-based models, using physically realistic representations of runoff processes and parameter values derived from small plots, to represent hydrology over a range of catchment complexity and scales (Connolly *et al.* 1990).

Apart from this, ANSWERS has not been widely used in Australia, although it has been applied to the Adelaide Hills Catchment in South Australia (McQuade *et al.* 1986) and on Whiteheads Creek in the Warragamba Dam catchment in NSW (Armstrong 1995). The large data requirements needed to run the model are likely to limit its application to Australian catchments in the future.

Examples of Model Users:
Queensland Department of Primary Industries

Hardware Requirements:
UNIX

Availability/Cost:
Distributed by C Vision Pty Ltd, 185 Elizabeth St Suite 320, Sydney NSW 2000, Australia. Tel: (02) 9283 4000; Fax: (02) 9261 4854.

3.2.4 AQUALM

AQUALM is a stormwater quality model that generates point and non-point source pollutants through standard or user-defined equations in addition to runoff estimations and routing (Phillips *et al.* 1993). AQUALM is a conceptual model similar to HSPF in the way that the catchment is divided into subcatchments with water and pollutants being routed between these subcatchments, although it is simpler than HSPF, using a lesser range of processes (Walton and Hunter 1996). The model has a rainfall-runoff component in addition to a pollution export component and can simulate the moisture storage characteristics for different land uses and runoff and pollutant export on a daily basis through the consideration of daily rainfall evaporation and soil infiltration (WBM-SKM 1997).

AQUALM consists of five modules: a daily rainfall-runoff module, point source and non-point source pollutant export modules, Best Management Practices (BMP) modules for sediment traps, gross pollutant traps, ponds, wetlands and lakes, a river quality and loading module, and a graphical user interface with an embedded decision support system (Phillips *et al.* 1993). The daily rainfall-runoff is simulated using a modified version of Boughton's model and predicts runoff from rainfall, accounting for interception, evapotranspiration and surface soil moisture storages.

The pollutant export module supports the simultaneous estimation of pollutant loads for up to ten pollutants, calculating non-point source pollutant loads from a subcatchment, pollutant inputs from a point source, and direct input of time-varying runoff and pollutant loads (Phillips *et al.* 1993).

The river quality and loading module is based upon a gradually varying 'conservation of mass flow' type model and incorporates user-defined decay functions to account for loss in constituent mass with flow downstream (Phillips *et al.* 1993).

Phillips *et al.* (1993) noted that a major impediment to the use of complex water quality models, such as SWMM or HSPF in their complete form, was the common lack of data on which to calibrate the model and the apparent large variability in the data that is available. AQUALM incorporates a number of well-tested water quality models with limited data requirements (Phillips *et al.* 1993).

The model requires the calibration of the rainfall-runoff and water quality components. Generally, the rainfall-runoff calibration is performed for each of the land use types featured within the catchment based on long-term rainfall and runoff data. However, often there is a lack of appropriate water quality data and therefore the water quality component of the model tends to be based upon characteristic export rates and data from other areas.

AQUALM has been used widely throughout Australia, mainly by government agencies. Two recent examples are in the review of the Sydney Water Proposal (1997) and the investigation of nutrients from urban stormwater and local water quality on behalf of the Goulburn Broken Water Quality Working Group (Anon. 1995). However, there have been few applications of AQUALM within the scientific literature, either in Australia or overseas.

Walden and Brodie (1995) identified that the AQUALM model had advantages over other models for pollutant export modelling for the Trinity catchment. Reasons included the ability to allow the application of generalised model coefficients based on previous experience, the flexibility of the model to investigate a number of land use scenarios, and the ability of the model to incorporate point source loading from sewerage treatment plants.

Examples of Model Users:
Goulburn Broken Water Quality Working Group; Sydney Water, Oxley Creek Co-ordinating Committee (part of Brisbane River Management Group); Sinclair Knight Merz (as part of Trinity Inlet Management Program)

3.2.5 CMSS

The Catchment Management Support System (CMSS) is a simple catchment-scale empirical model developed by CSIRO Land and Water to analyse the likely impacts of land use and land management policies on the nutrient load delivered to rivers, in particular the effect on total phosphorus and total nitrogen loads reaching waterways within a catchment.

The model is broken down into four modules: a database module, a policy module, a predictive model module, and an interrogation module (Davis and Farley 1997). With these, CMSS is able to account for the effects of land use and land management policies on nutrient loads. It calculates the contribution of different forms of land use to nutrient loads and allows the user to review the load and cost predictions.

The database module in CMSS contains four main files describing land uses, spatial attributes of the catchment, nutrient generation rates and management practices. The land use file describes the size of activities occurring within the catchment, generating a particular nutrient load per unit of the activity. Both point and non-point pollutant activities can be specified in this file. Areas of the catchment with the same type of land use but with differing environmental factors such as rainfall or slope gradient must be described as separate land uses within the CMSS structure.

CMSS calculates the average annual nutrient yield for a catchment using nutrient generation rates specified for each land use in the land use file. The total load for nutrient j is calculated as

$$Load_j = \sum_{i=1}^{n} \sum_{k=1}^{m} A_{ik} N_i g_{ij} \qquad (3.7)$$

where there are n land uses and m spatial units in the catchment. A_{ik} is the area of land use i in spatial unit k in the case of a diffuse source, or a count of the number of occurrences in spatial unit k for a point source. N_i is set to 1 for diffuse sources and is set to the catchment-averaged size of each occurrence for point sources. The term g_{ij} is the generation rate of nutrient j for land use i.

The nutrient generation rates must be obtained through either local or expert knowledge of the catchment, or from previous model application. Appendix I looks in more detail at nutrient export rates in Australia as given in the literature. Nutrient generation in the model is independent of rainfall events within the catchment. CMSS does not attempt to model processes such as rainfall-runoff or infiltration. The model is also capable of assessing the most important sources of nutrient load within the catchment and performing cost–benefit analysis on various pollution policies.

CMSS produces results in the form of a GIS-like map that allows the model user to locate land uses and land features in the catchment that produces the most nutrients, which can then be used to design and implement appropriate management practices. CMSS is easier to use and has fewer data requirements than conceptual or physics-based models, but does not generally give as accurate an assessment of water quality as models of these types according to Walton and Hunter 1996. As such, CMSS is most useful as an initial planning tool to give relative rankings of catchments and land uses with respect to nutrient loads.

CMSS has been widely used throughout Australia since its development, with 60 registered users throughout Australia and New Zealand by 1997 (Hook 1997). Due to the simplicity of the model, CMSS can be used by catchment groups and government agencies with limited data or technical knowledge about the processes involved in water quality issues. In particular, the model has been used extensively by the NSW Algal Management Strategy (Long and Verhoeven 1995; Verhoeven 1995; Porter and Foster 1996).

Examples of Model Users:
NSW Algal Management Strategy (Barwon region of NSW); Mount Lofty Ranges Review Team (South Australia)

Hardware Requirements:
PC

Availability/Cost:
$1300 including 3-day training course, user manual, nutrient data book, and an expert system for estimating nutrient generation rates (Hook 1997).

For further information contact:
Dr. Bill Young
Phone: (02) 6246 5729
Email: wjy@cbr.clw.csiro.au

3.2.6 CREAMS

The Chemical Runoff and Erosion from Agricultural Management Systems model (CREAMS) was developed as a tool to evaluate the relative effects of agricultural practices on pollutants in surface runoff and in soil water below the root zone (Knisel 1980; Lane *et al.* 1992; Lane *et al.* 1995). The model has been extended and modified in GLEAMS, the Groundwater Loading Effects of Agricultural Management Systems (Ball and Trudgill 1995). Both of the models consist of three components: hydrology, erosion/sedimentation, and chemistry. The model predicts erosion, deposition and transport of sediment on an overland flow slope profile and into first- and second-order channels (Silburn and Loch 1991). CREAMS applies to field-sized catchments, assumed to be uniform in soil topography and land use, of approximately 40 ha, although it can be used on scales up to 400 ha (Lane *et al.* 1992).

The CREAMS model uses a physics-based approach to erosion and sediment transport, although significant simplifications are made as a consequence of model size, computational speed and limited hydrologic data. The erosional model is run for individual storms and assumes quasi-steady state through the use of a characteristic runoff rate for each storm (Silburn and Loch 1989). Additionally, slope is assumed to be uniform and is computed on a per-unit-width basis.

Unlike many other models, such as USLE, CREAMS considers that the amount of sediment leaving a field is limited by transport capacity or detachment. In addition, CREAMS also considers gully erosion which, although not considered by USLE, can produce as much sediment as that produced by sheet and rill erosion (Lane *et al.* 1992). Sediment transport is calculated according to the steady-state continuity equation

$$\frac{dG}{dx} = D_f + D_s \tag{3.8}$$

where G is the sediment load, x is the distance, D_f is the detachment or deposition rate by flow and D_s is the rate that sediment is added to the flow from lateral areas.

Rainfall-runoff in the CREAMS model is simulated using the SCS curve number approach as described in section 3.2.2 in the AGNPS model. Sediment yield, as with the ANSWERS model, is calculated using Yalins' equation.

The CREAMS model has started to be used more regularly within Australia for soil erosion prediction over the last ten years. In particular, Silburn and Loch (Loch *et al.* 1989a; Silburn and Loch 1989; Silburn and Loch 1991) have assessed the validity of the model's predictions in laboratory and field studies. These studies have generally indicated that the performance of the model is acceptable, although Evans *et al.* (1994, 1997) identified that the interrill component of CREAMS may be oversensitive to slope.

As noted previously, CREAMS accounts for gully erosion and deposition, unlike models such as USLE. Additionally, the model allows for the erodibility factor to be updated from one runoff event to the next (Govers and Loch 1993). As soil erodibility factors have been shown to be quite variable, this could be an important aspect of the model. However, Govers and Lock (1993) noted that the 'dynamic nature of runoff erosion may limit any increase in prediction accuracy that can be obtained using physics-based models rather than statistical models, as the performance of a model such as CREAMS will become highly dependent on the accuracy of the input data'. Another potential disadvantage of the CREAMS model is that the plot or catchment being modelled is assumed to be uniform in soil topography and land use, a highly unrealistic assumption. In other words, the benefits associated with the consideration of gully erosion and deposition processes may be nullified by the dependency of the model on data accuracy and on assumptions of homogeneity.

Examples of Model Users:
Land Management Research Branch, Queensland Department of Primary Industries, Toowoomba, Queensland.

Hardware Requirements:
UNIX (using Fortran)

3.2.7 HSPF

The Hydrologic Simulation Program – Fortran (HSPF) is a model developed upon the 1960s Stanford Watershed Model (WBM-SKM 1997) for the simulation of watershed hydrology and water quality (N, P, SS and other toxic organic or inorganic pollutants) (Walton and Hunter 1996). HSPF considers only one-dimensional flow and is suitable for non-tidal reaches of rivers. The model is a catchment-scale, grid-based, conceptual model whereby catchments are broken down into subcatchments and water quantity and quality are calculated for each land use within the subcatchment. Water, sediment and chemical fluxes are then added to the streams, and flows are routed to the catchment outlet.

The model consists of three main modules: the pervious land module, the impervious land module and the river / mixed reservoir module. In the pervious land module, hydrologic processes are driven by rainfall and include interception of rainfall, evaporation, overland flow, infiltration, interflow, soil moisture storage and groundwater (Cheung and Fisher 1995). Surface erosion is accounted for by the processes of detachment and transport, although dust deposition and wind-blown removal can also be simulated. Sediment-adsorbed water quality components are treated as being washed off with sediments and entering the receiving stream (Cheung and Fisher 1995).

The impervious land module is simpler than the pervious module, with dissolved solutes and accumulated sediments being transported off the land surface with overland flow. Sediment-adsorbed water quality components are treated as with the pervious land module. Urban areas that consist of pervious and impervious surfaces are modelled by assigning a portion of the land as impervious and the remaining land according to the make-up of the land (Cheung and Fisher 1995).

The river and mixed reservoir module includes physical processes such as transport advection, diffusion, sediment deposition and scouring. The model also considers the following chemical processes: aeration, nitrification, denitrification, biochemical oxidation, adsorption and desorption of solute from suspended sediment, and settlement (Cheung and Fisher 1995). Chapman (1991) tables the specific transport and reaction, as well as general, characteristics of HSPF and other toxicant models.

The inputs to the model include rainfall, evaporation, air and water temperature, solar radiation, sediment grain size distribution, point source discharge volume, and water quality data (Cheung and Fisher 1995). Streamflow and in-stream water quality variables are used for comparison with the model results. The model is able to simulate a wide range of water quality components. The outputs from the simulation are a temporal history of runoff flow rate, sediment load and nutrient concentrations along with a temporal history of water quantity and quality at any point in the catchment.

HSPF was developed as a generic model designed to apply to most catchments using existing meteorological and hydrological data, soils and topographic information, and information on drainage and other characteristics (Rahman and Salbe 1993). A limitation to this model is that it relies heavily on calibration against field data for parameterisation (Walton and Hunter 1996). With the relatively large number of parameters required to be calibrated this raises problems associated with parameter identifiability and the physical meaningfulness of model parameters. Although HSPF has the potential to be a useful tool for catchment management, Cheung and Fisher (1995) note that the calibre of models found is related to the availability and accuracy of input data and the skills of the modeller.

HSPF has been used widely in southern Australia, particularly by Sydney Water and the Australian Water Technologies Science and Environment Division in NSW. Particular applications of the model have included the upper Nepean catchment, (Ball *et al.* 1993), the South Creek catchment, (Fisher *et al.* 1993, Rahman and Salbe 1993), the Werriberri catchment (Cheung 1993; Fisher and Deen 1993; Cheung and Fisher 1995) and the Cattai catchment (Cheung and Fisher 1995). The HSPF model was used because of the identified comprehensiveness and flexibility of the model and its ability to simulate runoff and in-stream process simultaneously (Rahman and Salbe 1993; Cheung and Fisher 1995).

Examples of Model Users:
Queensland Department of Primary Industries, Resource Management Institute; Australian Water Technologies Ensight / Sydney Water Corporation.

Hardware Requirements:
PC (DOS) or UNIX

Availability/Cost:
Available for DOS systems from: ftp.epa.gov/epa_ceam/ or http://www.cee.odu.edu/cee/model/hspf.html (program and user manual)

3.2.8 HYDRA

HYDRA is a collaborative project between CSIRO Land and Water, CSIRO Mathematical and Information Services and Sydney Water (Rizzoli and Young 1997). The aim of this project is to integrate current modelling approaches into an Environmental Decision Support System (EDSS). The EDSS was designed for planning water quality management in the Hawkesbury–Nepean and combines hydrological modelling systems with a map interface easily used by planners and managers (http://www.dit.csiro.au/hydra.htm). HYDRA uses a GIS-based interface to allow the user to depict land use changes. Predicted outcomes are examined by clicking on a particular section and generating a chart of the water quality measure (e.g. nutrients and sediment) over the period of interest.

Examples of Model Users:
Sydney Water Corporation

Availability/cost:
Not commercially available yet

For further information contact:
Susan Cuddy
Phone: (02) 6246 5705
Email: susan.cuddy@cbr.clw.csiro.au

3.2.9 IHACRES

See STARS, 3.2.15.

3.2.10 IQQM

The Integrated Water Quantity and Quality Model (IQQM) is a conceptual model being developed by the Department of Land and Water Conservation NSW. IQQM has modules for in-stream water quality and quantity as well as for rainfall-runoff and groundwater quantity (DLWC 1995; Simons *et al.* 1996). IQQM operates on a continuous basis, using time steps of one day, down to one hour for some processes.

The main processes that are simulated in the instream water quantity module include flow routing in rivers, effluent systems and irrigation channels, reservoir operation, irrigation, urban water supply and other consumptive uses, and wetland and environmental flow requirements. The instream water quality module is based on the program QUAL2E, developed by the USEPA, and accounts for factors such as nitrogen, phosphorus, dissolved oxygen and sediment, as well as coliforms and algae. IQQM also has a module designed to simulate salt mobilisation in catchments where the major source of salt is rock weathering.

Examples of Model Users:
The model has been used by the Department of Land and Water Conservation in the Border Rivers, Barwon–Darling between Mungindi and upstream of Menindee lakes, Macquarie, Lachlan, Clarence, Namoi, Hunter, Gwydir and Murrumbidgee river systems (Hook 1997).

Hardware Requirements:
8 MB RAM, 80386 CPU with maths coprocessor, 40 MB hard disk, SVGA monitor (DLWC 1995).

For further information contact:
Dr Dugald Black
Phone: (02) 9895 7421
Email: dblack@dlwc.nsw.gov.au

3.2.11 LASCAM

LASCAM, a salt and water balance model, has been adapted to include a sediment generation and transport algorithm for modelling hydrological processes at a catchment scale. Viney and Sivapalan (1997) incorporated a conceptualisation of the Universal Soil Loss Equation to predict sediment generation, E, according to the equation:

$$E = \gamma C \, q_{ie}^{\delta} \tag{3.9}$$

where q_{ie} is the daily infiltration-excess runoff, C is the USLE crop factor and the variables δ and γ are optimisable parameters. The model has been used extensively in the Swan–Avon River Basin in Western Australia to predict and model sediment loads, water yields, salinity and nutrients (Viney and Sivapalan 1997; http://www.cwr.uwa.edu.au/index2.html). The Centre for Water Research at the University of Western Australia has undertaken much of the work involving this model.

LASCAM-WQ is a conceptual model adapted from LASCAM and uses gridded topographic information to define a stream network and break up the catchment into a series of subcatchments (Viney and Sivapalan 1997). The hydrological processes are modelled at the subcatchment scale before being summed up to represent the total catchment. The rainfall-runoff component of the model contains 22 parameters.

LASCAM requires daily rainfall, pan evaporation and land use information while topographic data is required to define subcatchments and the stream network. The outputs for the model are surface and subsurface runoff, actual evaporation, recharge to the permanent groundwater table baseflow, measures of soil moisture and salt outflows (http://www.cwr.uwa.edu.au/index2.html).

The conceptual nature of LASCAM-WQ requires that the model be calibrated against a long time series of measured streamflow and water quality data. The model can use measurements of water outflow, salinity, nutrients and sediments at one or more locations within the stream catchments network. The model has shown considerable potential as a sediment yield model (Viney and Sivapalan 1997) and has predicted water yield, salinity, sediments, nitrogen and phosphorus for the entire Swan–Avon River Basin (http://www.cwr.uwa.edu.au/index2.html). Despite the need for calibration, LASCAM can potentially provide an advantage over the use of physics-based sediment models, given the considerable data and parameter uncertainties identified by Viney and Sivapalan (1997). The smaller number of parameters needed to be calibrated for the water quality component means that this part of the model is less likely to suffer from problems associated with identifiability than other more complex models.

Examples of Model Users:
Centre for Water Research at the University of Western Australia

Availability/Cost:
Not commercially available yet
For further information contact:
Assoc. Prof. M. Sivapalan
Phone: (08) 9380 2320
Email: sivapala@cwr.uwa.edu.au

3.2.12 LISEM

The Limburg Soil Erosion Model (LISEM) is a physics-based hydrological and soil erosion model developed by the Department of Physical Geography at Utrecht University and the Soil Physics Division at the Winard Staring Centre in Wageningen, the Netherlands, for planning and conservation purposes. The LISEM model is based upon EUROSEM. LISEM is completely incorporated within a GIS, that is, the model is expressed completely in the GIS command structure of PCRaster.

The LISEM model does not simulate concentrated erosion in rills and gullies; rather, it simulates flow detachment in the ponded area only. This can be seen as an intermediate between sheet and rill erosion.

LISEM incorporates a number of different processes including rainfall interception, surface storage in micro-depressions, infiltration, vertical movement of water in soil, overland flow, channel flow, detachment by rainfall and throughfall, detachment by overland flow and transport capacity of flow. Model simulation is based on the solution of a number of physical equations describing water and sediment yield processes. LISEM simulates the runoff and sediment transport caused by a single rainfall event.

The GIS nature of LISEM means that inputs to the model simulation are in the form of GIS maps. Approximately 25 maps are required for simulation, including maps describing catchment morphology, maps required by the soil water submodel and maps with soil and land use inputs. Rainfall data from multiple rainfall gauges must also be input. LISEM generates from this a map showing the spatial distribution of rainfall intensity. Thus LISEM incorporates both the spatial and temporal variability of rainfall.

Outputs of the LISEM model include totals for such variables as runoff, sediment, infiltration and storage depression. Maps showing the spatial distribution of such factors as soil erosion and deposition, and maps of overland flow at desired time intervals during the simulation are also produced by LISEM. The model is also capable of producing hydrographs and sediment graphs for a rainfall event simulation.

The model has not been used much in Australia, having been developed for the Dutch region of South Limburg (Takken *et al.,* no. 87, in press). Given the likely differences in climate and soil type between NSW and the area for which the model was developed, detailed validation and testing of the model will be required prior to use in Australia. The detailed representation of LISEM, even though linked to a GIS, is likely to limit the application of LISEM, or similar models, except for large detailed research projects on fairly small catchments. The LISEM model requires detailed spatially and temporally variable data which has limited availability, especially in Australian catchments. Like most other physics-based models, LISEM can be expected to suffer from difficulties associated with model identifiability and data availability.

Examples of Model Users:
No Australian applications identified.

3.2.13 MIKE-11

MIKE-11 is a software system used for water quality modelling developed by the Danish Hydrologic Institute (DHI). The model is a one-dimensional (cross-sectionally averaged) dynamic model consisting of a number of modules (Hanley *et al.* 1998). The basic modules are a rainfall-runoff component, a hydrodynamic module, a water quality module, and a sediment transport module. MIKE-11 simulates flow using St Venant's complete non-linear equations of open channel flow that can be solved numerically between all points at specified time intervals for given boundary conditions.

The advection–dispersion module is based on the one-dimensional equation of conservation of mass of dissolved or suspended materials and includes a description of the erosion and deposition of cohesive sediment.

The water quality module simulates the reaction processes including the degradation of organic matter, photosynthesis and respiration of plants, nitrification and the exchange of oxygen with the atmosphere.

The model simulates unsteady one-dimensional flows and accounts for the interdependence of sediment transport, alluvial roughness and hydrodynamics in the simulation of equilibrium conditions of the river; a capacity essential in determining morphological changes and erosion patterns associated with mining operations (Kwan and Abbey 1993). However, the model is limited by its one-dimensional representation of processes. It neglects secondary currents and ignores bank erosion processes. Additionally, the model requires data to define temporal and geometric boundary conditions and data for development, calibration and testing of the model.

The accuracy of the MIKE-11 model is undermined by a number of factors. The first of these is the use of one-dimensional equations to represent three-dimensional processes. Many of the important interactions within the system are ignored or simplified in this process. This raises questions about the physical interpretability of the model. The large data requirements of the model mean that the model is likely to suffer from problems caused by error accumulation, or from a lack of identifiability of model parameters in situations where model parameters must be calibrated. The justifiability of using measured physical parameters within the model given the oversimplification of physical processes inherent in a one-dimensional representation of the physics of the catchment system is also questionable. Thus the additional complexity of model calculations does not seem warranted, given that these factors undermine any additional accuracy.

The MIKE-11 model has been used extensively in Australia within government agencies, consultancy firms and universities, particularly in southern Australia. Much of the emphasis has been placed on morphological modelling (e.g. Kwan and Abbey 1993), hydraulic modelling (Bernard 1993) and preliminary water quality modelling (Western *et al.* 1993)

Examples of Model Users:
CRC for Catchment Hydrology, Melbourne; Murray–Darling Basin Commission; Queensland Department of Primary Industries; Sydney Water Corporation; University of Melbourne; NSW Public Works; Sinclair Knight Merz

For a more comprehensive list of installations, go to:
http://www.dhi.dk/mike11/M11List.htm

Hardware Requirements:
DOS, UNIX

Availability/Cost:
Distributed by C Vision Pty Ltd
185 Elizabeth St Suite 320, Sydney NSW 2000
Tel: (02) 9283 4000
Fax:(02) 9261 4854.

3.2.14 PERFECT

The Productivity, Erosion and Runoff, Functions to Evaluate Conservation Techniques (PERFECT) model was developed by the Queensland Department of Primary Industries (Land Management Branch, Queensland Wheat Research Institute) and the QDPI/CSIRO Agricultural Production System Research Unit (Littleboy *et al.* 1992b). The model was developed in response to the limited applicability of models such as CREAMS for analysing the effects of soil management practices such as tillage or fallow management strategies (Littleboy *et al.* 1996). Models such as CREAMS calculate runoff as a function of rainfall and soil water content, excluding surface and crop cover changes resulting from tillage practices. PERFECT was designed to predict runoff, erosion and crop yield for some management options in dryland cropping areas of Australia, including sequences of plantings, harvests and stubble management during fallows (Littleboy *et al.* 1996).

The model comprises six modules: data input, water balance, crop growth, crop residue, erosion and model output. These modules are arranged in a framework that allows alternative modules to be used as required for the potential range of applications. The modules draw on other models such as MUSLE and CREAMS. Erosion is simulated in the model using MUSLE, while the mineral nitrogen removed from the topsoil by erosion is simulated using the following relationship taken from CREAMS:

$$SEDN = SOIL \times MNIT \times ENR \tag{3.10}$$

where *SEDN* is the mineral nitrogen lost in the sediment, *SOIL* is the daily erosion (kg ha^{-1}), *MNIT* is the mineral nitrogen in the topsoil (kg kg^{-1}) and *ENR* is the enrichment ratio (Littleboy *et al.* 1992b).

The inputs to the models are daily climate data, soil parameters, cropping sequence criteria (i.e. crop type and length of fallow), crop growth parameters and fallow management (tillage) options. The climate data requirements include daily rainfall, pan evaporation, temperature and evaporation.

Littleboy *et al.* (1992b) found that PERFECT was more reliable than CREAMS in predicting runoff, accounting for 77%–89% of the variation in daily runoff volume. This, in addition to the consideration of crop cover and surface runoff on infiltration and soil evaporation, indicates that PERFECT is a more appropriate model to analyse runoff from cropping systems with complex crop/fallow rotations than the CREAMS model. Although PERFECT was not developed specifically as a water quality model, the incorporation of a runoff component in addition to the large crop component of the model may provide an advantage over models such as CREAMS, where the major emphasis is placed on surface hydrology, sediment and pesticide movement, and nutrient models with little or no accounting for land management practices.

The major disadvantage with PERFECT in terms of water quality and erosion modelling is that it does not contain a sediment transport or nutrient component. However, the structure of the model is such that a hydrological component of the model may be incorporated. Additionally, the erosion component of the model does not account for rainfall intensity, thus raising the possibility for overestimation or underestimation of erosion depending on the rainfall event. Although the model structure is generally robust, Littleboy *et al.* (1992a) noted that the model was not designed for application beyond those environments typical of north-eastern Australia and recommended that the model be calibrated against suitable field data before use in any other environment.

In summary, PERFECT provides a potentially valuable tool for assessing conservation cropping options by simulating the water balance, crop yield and erosion for combinations of soil type, climate, fallow management strategy and cropping sequence. The incorporation of a sediment transport and nutrient component would be required for the model to be useful in water quality modelling. If this were to occur, the detail of the crop cover and management components may provide an advantage over other models, where these processes are considered important.

Examples of Model Users:
Queensland Department of Primary Industries

Hardware Requirements:
IBM PC, UNIX

Availability/Cost:
The model, including source code, is available free of charge.

For further information contact:
Dr. Mark Littleboy
Phone: (07) 3896 9593
Email: markl@salt.ind.dpi.qld.gov.au

3.2.15 STARS and IHACRES

The Solute Transport with Advection, Resuspension and Settling (STARS) model was developed at the Integrated Catchment Assessment and Management Centre at the Australian National University. It is a one-dimensional model of advective transport between two gauging stations or nodes given flow at both nodes (Green *et al.* 1999; Dietrich *et al.* 1999).

The STARS model is conceptually based, and requires upstream and downstream concentration over some period (including a few events) for calibration of the model parameters. The model has only five parameters and is thus less likely to experience problems with model identifiability than more complex conceptual and physics-based models.

The model simulates processes such as particle settling, deposition and resuspension of sediment as well as lateral sources of sediment from bank erosion and sediment inputs associated with local rainfall. The model compensates for differences in flow between upstream and downstream nodes by computing an average flow rate over the reach, Q_t. The scaled equation for downstream suspended sediment concentration, c_L, as a function of time is:

$$c_L(t) = c_0(t-\tau)e^{-\frac{\alpha}{Q_t}} + \left\{\frac{\beta\eta(Q_t - Q_*)^{\mu} + \gamma}{\alpha}\right\}\left\{1 - e^{-\frac{\alpha}{Q_t}}\right\} \tag{3.11}$$

where c_0 is the concentration upstream, τ is the effective water parcel travel time estimated from the data, and

$$\eta = \begin{cases} 1 & Q_t > Q_* \\ 0 & otherwise \end{cases} \tag{3.12}$$

Deposition is controlled by the particle settling velocity via α, lateral sources are given by γ and resuspension is determined by a combination of β, Q_* and μ. These are the five parameters requiring calibration in the model.

When it is necessary to model streamflow, the IHACRES model is used (Jakeman *et al.* 1994a, 1994b, 1990; Evans and Jakeman 1997) for predicting discharge at catchment outlets, and a simple discharge routing model is used for instream sections. The IHACRES rainfall-runoff model is a hybrid metric–conceptual model based on the instantaneous unit hydrograph. It was developed by the Centre for Resource and Environmental Studies with the Institute of Hydrology. This model accounts for the effects of evapotranspiration, drainage and precipitation on rainfall-runoff. Rainfall is modified using temperature data to reflect the effects of drainage, evapotranspiration and antecedent weather conditions to become effective rainfall. This effective rainfall is then modelled as passing through one or two internal reservoirs or storages. The exact number of storages used is determined by the calibration data. IHACRES has been widely applied within Australia and overseas in a range of climatic conditions. It has been shown to predict runoff as effectively as other models but has the advantage of containing only 5–7 parameters. It has been augmented with power law relations between sediment concentrations and discharge (and between phosphorus and sediment concentrations) to predict water quality concentrations. This has been successfully prototyped in several catchments of the Namoi Basin (Jakeman *et al.* 1999).

The STARS and IHACRES models have the advantage of requiring relatively little input data, as the conceptual nature of the models means that spatially distributed input data on catchment characteristics is not required for model calibration. The small number of model parameters also means that the models are less likely to suffer from problems of identifiability than more complex models.

STARS and IHACRES were developed in Australia, and as such are applicable to Australian conditions. STARS has been applied to catchments in the Namoi, Murrumbidgee and Murray River Basins (Green *et al.* 1997; Dietrich and Jakeman 1997).

Examples of Model Users:
Centre for Resource and Environmental Studies (CRES), Integrated Catchment Assessment and Management Centre (ICAM).

Availability/Cost:
The STARS model is not commercially available. IHACRES is available for approximately £300 from:

Institute of Hydrology, Maclean Building

Wallingford OX10 8BB UK
Phone: +44 1492 83 8800
For further information contact:
Professor Tony Jakeman
Phone: (02) 6249 4742
Email: tony@cres.anu.edu.au

3.2.16 THALES

THALES is a hydrological model that uses TAPES-C, a set of computer programs that allypartitions automatically subdivides the model area into interconnected irregular-shaped elements and calculates a number of topographic attributes for each element (Moore and Grayson 1993). The THALES model is event-based and models runoff and subsurface flow (Hatton *et al.* 1998). The model has the potential to be incorporated into a sediment and nutrient transport model where the simulated flow characteristics of the catchment would be used to calculate soil movement or nutrient transport (Grayson and Moore 1993).

THALES is a relatively simple physics-based model that enables a wide range of hydrologic processes to be represented through the incorporation of the Hortonian mechanism of surface runoff as well as a representation of variable-source-area runoff and exfiltration of subsurface flow (Grayson *et al.* 1992a). The elemental structure of THALES allows each element to have different infiltration, surface flow and subsurface flow parameters, although parameters are generally measured for each soil type or region of different surface conditions and it is assumed that these do not vary within each region or soil type. Grayson *et al.* (1992b) note that assumptions underlying models such as THALES are extensive and occur at all levels from the overall model structure to the constituent algorithms. As a consequence, there is a danger in using this model out of context.

THALES has been developed as an investigative tool and has proved useful in the analysis of catchment response. Moore *et al.* (1991) stated that a common deficiency of many hydrologic or water quality models is their inability to represent the effects of three-dimensional terrain on flow processes without a large number of often unrealistic assumptions. In this way, THALES, with the use of TAPES-C, provides an advantage over models that do not account for three-dimensional terrain. At present, THALES has been used mainly in research and will require further development to incorporate a water quality component prior to use by catchment managers.

Examples of Model Users:
Department of Civil Engineering, University of Melbourne

Hardware Requirements:
DEM component (TAPES) requires UNIX with X Windows graphics (written in Fortran-77 and C). Not compiled under DOS or Windows.

Availability/Cost:

TAPES-C, THALES, and TAPES-G (grid-based version of TAPES) available from http://cres.anu.edu.au/software/tapes.html.

For further information contact:

Dr. Rodger Grayson
Phone: (03) 9344 7305
Email: rodger@civag.unimelb.edu.au

3.2.17 USLE and modifications

The Universal Soil Loss Equation (USLE) is a soil erosion prediction model used widely within the USA and worldwide, either on its own or incorporated into such models as AGNPS. Developed in the 1970s by the US Department of Agriculture, the model has undergone much research and a number of modifications (e.g. MUSLE, USLE-M). The model has also been upgraded to take into account additional information that has become available since the development of the USLE (RUSLE). The basic USLE is an empirical overland flow or sheet–rill erosion regression equation based primarily on observations (Zhang *et al.* 1995). Although USLE is an empirical model, it has some conceptual components. The model relates sediment delivery to slope, slope length, rainfall, erosivity and soil erodibility, of which the latter two are predicted both empirically and conceptually.

The USLE estimates the average annual soil loss from:

$$A = R\,K\,L\,C\,S\,P \tag{3.13}$$

where A is the soil loss averaged over slope length, R is the combined erosivity of rainfall and runoff, K is the soil erodibility, L is the factor dependent on slope length, S is the factor dependent on slope gradient, C is dependent on vegetative cover and management and P is dependent on conservation practices (Zhang *et al*. 1995). The simplicity of this equation and the availability of parameter values, at least in the USA, has made this model relatively easy to use (Loch and Rosewell 1992).

There are a number of limitations to the USLE equation. The model is not event-based and as such cannot identify those events most likely to result in large-scale erosion. Gully erosion and mass movement are not considered in the erosion process, and the deposition of the sediment is not considered to occur within the area under consideration (Zhang *et al*. 1995). Runoff leaving a field generally concentrates in a few major channels, the profile of which is often concave, such that ephemeral gully erosion can occur along the upper reach of the channel and deposition occurs in the lower reaches of the channels. This gully erosion can be as extensive as sheet and rill erosion (Lane *et al.* 1992). Additionally, unlike in the USA, the use of USLE in Australia has been limited by the perceived lack of data for the parameters required to run the model under Australian conditions (Loch and Rosewell 1992). Nearing *et al.* (1994) noted that the adaptation of USLE to a new environment requires a large investment of time and resources to develop the database required to run the model. Evans *et al.* (1992) identified that due to rainfall variability, data must be collected for at least 10 years and this, combined with the lack of data for overburden spoil and replaced spoils to be applied to USLE, was a disadvantage for the use of this model in spoil pile erosion prediction.

Due to the identified limitations of USLE, a number of modifications to the basic format for have been proposed in the literature. These include the Modified USLE, the Revised USLE (Renard and Ferreira 1993; Renard *et al.* 1994), the USLE-M (Kinnell 1998a; Kinnell 1998b; Kinnell and Risse 1998) and SOILOSS (Rosewell 1995; Rosewell and Lang 1996). These continue to improve components of the model, tending to make it more process-based. RUSLE maintains the basic form of the USLE, although all equations used to arrive at the factor values have been modified (Lane *et al.* 1992). Changes to the form of the LS factor in RUSLE enables the prediction of soil loss due to Hortonian overland flow in three-dimensional terrain with convergent and divergent slopes (Ryan and McKenzie 1997). USLE-M, for example, provides a more complex representation of processes than the USLE as it more directly considers the effect of runoff on erosion with changes to the R factor (Kinnell 1998b). Consequently, USLE-M has a greater ability to account for the more frequent small to medium erosion losses. Kinnell (1998a) noted that ‘USLE technology will form the basis of modelling the spatial and temporal variation in soil erosion within catchments in the future and as such there are benefits in continued improvements in the model’.

SOILOSS

The SOILOSS computer program is a local adaptation of RUSLE, being adapted to NSW conditions through the estimation of the rainfall erosivity and soil erodibility factors from local rainfall erosivity maps or calculated from rainfall intensity data and soil landscape maps respectively (Rosewell and Lang 1996). The map required for the estimation of rainfall erosivity factors can be obtained from Rosewell and Turner (1992). The program applies the USLE and is used to assist in the selection of land and crop management practices to decrease erosion (Rosewell 1995). The SOILOSS program has been used extensively by the Soil Conservation Service, now the NSW Department of Land and Water Conservation, to estimate water pollution hazard for Water Pollution Requirements (Rosewell and Lang 1996). This was achieved by combining the site-specific factors of R, K and S for a fixed slope length of 20 m and a P factor of 1. The cover management factor, C, is calculated based upon average soil loss levels following specific logging operations (Rosewell and Lang 1996). Factor C measures the combined effect of all the interrelated cover and management variables and is defined as the ratio of soil loss from land managed under specified conditions to the corresponding loss from clean-tilled continuous fallow (Rosewell 1997).

The main advantage of RUSLE, on which SOILOSS is based, over the USLE is that the RUSLE has the capacity to estimate the *C* factor from information on vegetation form, decay and tillage practices rather than from experimental plot data as used in the USLE.

Another advantage of SOILOSS over other USLE-based alternatives is that it is applicable to Australian conditions and should thus be more reliable in erosion predictions. On the other hand, the SOILOSS program is still a non-event-based prediction equation (Kinnell 1996), and as discussed previously may not be as useful a management tool as an event-based predictive model. However, Rosewell (1995) noted that a combination of SOILOSS with AGNPS is capable of indicating the relative differences in nutrient generation between alternative land and crop management practices. The incorporation of SOILOSS into models in place of USLE would be expected to improve the validity of the model predictions under Australian conditions.

Examples of Model Users (SOILOSS):
NSW Department of Land and Water Conservation; NSW Department of Housing; NSW Environment Protection Authority; State Forests of NSW; NSW Agriculture

Availability/Cost (SOILOSS):
Available from Publication Sales, Department of Land and Water Conservation, GPO Box 39, Sydney, NSW 2001. (Cost in 1997 $100 for single user and $200 for multiple users.)

For further SOILOSS information contact:
Mr. C.J. Rosewell
Phone: (02) 6742 9505
Email: cdrose@ozemail.com.au

3.2.18 WEPP

The Watershed Erosion Prediction Project (WEPP) is a physics-based, hillslope-scale model developed in the USA in an initiative between the Agricultural Research Service, the Soil Conservation Service, the Forest Service in the Department of Agriculture and the Bureau of Land Management in the US Department of the Interior (Laflen *et al.* 1991). The model has been applied widely to hillslopes in the US (e.g. Laflen *et al.* 1991) and worldwide, including Australia (e.g. Fogarty 1997). The model was intended to determine and/or assess the essential mechanisms controlling erosion by water, including anthropogenic impacts (Zhang *et al.* 1995; Liu *et al.* 1997).

Like many physics-based models, WEPP is based on a mass balance formulation, one of the standard equations used in physics-based models:

$$\frac{dq_s}{dx} = D_r + D_i \qquad (3.14)$$

where $\frac{dq_s}{dx}$ is the sediment rate per unit width of rill channel, D_r is the rill net detachment or deposition rate, and D_i is the interrill net detachment or deposition rate (Zhang *et al.* 1995). Being a physics-based model, the computational requirements of WEPP are high, with a large number of inputs required. The processes represented by WEPP can be broadly characterised as erosional processes, hydrological processes, plant growth and residue processes, water use processes, hydraulic processes and soil processes (Laflen *et al.* 1991). Erosional processes are limited to sheet and rill erosion and erosion occurring in channels where detachment is due to hydraulic shear. Through the erosional components of the model, the three stages of erosion (detachment, transport and deposition) are quantified using the rill–interrill concept of describing sediment detachment (Laflen *et al.* 1991; Lane *et al.* 1995), which is the detachment and transport of sediment through raindrop impact and shallow flows.

Originally, interrill detachment was modelled in WEPP as:

$$D_i = K_i I^2 \tag{3.15}$$

where D_i is the interrill detachment rate, K_i is the interrill erodibility constant and I is the rainfall intensity. Following work by Kinnell (1993a, b), the I^2 term was replaced by the product of runoff and intensity in the 1995 release of WEPP.

Rill detachment is modelled using the relationship:

$$D_c = K_r(\tau - \tau_c) \tag{3.16}$$

where D_c is the detachment capacity of clear water, K_r is the rill erodibility of soil due to hydraulic shear, τ_c is the shear below which there is no detachment and τ is the hydraulic shear of flowing water, where

$$\tau = \gamma r_h s \tag{3.17}$$

and γ is the density of water, r_h is the hydraulic radius and s is the hydraulic gradient, which is approximately equal to the slope of the rill bottom.

The erosional processes result from the forces and energies developed in hydrologic processes (Laflen *et al.* 1991). The components of the hydrologic processes are climate, infiltration and a winter component that accounts for snow accumulation and melt.

Knowledge of plant growth and residue components is required to make an accurate assessment of the plant and residue characteristics above and below the soil. These include canopy cover and height, above- and below-ground biomass of living and dead plant material, leaf area index and basal area, and are estimated on a daily basis (Laflen *et al.* 1991). As such, information regarding dates and management practices are essential inputs to the model. The plant characteristics are of utmost importance to describe adequately as they will have a large impact on the soil erosion and hydrological processes within the site.

The soil water status is updated on a daily basis and is required to obtain infiltration and surface runoff volumes—the driving force in the detachment by flowing water in rills and channels (Laflen *et al.* 1991). The water balance component uses information about climate, plant growth and infiltration to estimate daily potential evapotranspiration and soil and plant evaporation.

The hydraulic processes component computes the hydraulic shearing forces exerted on the soil surface by the surface runoff. This requires information regarding surface runoff volumes, hydraulic roughness, and approximations of runoff duration and peak rate.

The final component of the model, the soil processes module, deals with the temporal changes in soil properties important in soil erosion, considering the effect of management practices, weathering, consolidation, and rainfall on soil and surface variables, including random roughness, bulk density, saturated hydraulic conductivity, and the erodibility factors of the rill and interrill (Laflen *et al.* 1991).

As can be seen from the above description, WEPP requires a large amount of input data. The outputs of the model can be summarised as spatial and temporal distributions of soil loss, sediment yield, sediment size characteristics, runoff volumes and soil water balance. The WEPP profile also considers sediment deposition and is applicable from the top of a hillslope to a channel.

Zhang *et al*. (1995) noted that the individual processes and components which affect erosion, including the complex interactions between various factors and their temporal variabilities, are simply and effectively described. In addition, the ability of WEPP to accurately predict where detachment and deposition will occur will be useful in establishing appropriate conservation or management practices.

There are a number of possible criticisms of the WEPP model. Firstly, the large computational requirements of the model may limit its applicability in studies of Australian catchments where there is often little data or available resources. Many of the model parameters may need to be calibrated against observed data in such studies, creating problems with model identifiability and the physical interpretability of model parameters. Secondly, the model was developed for the hillslope scale. With increasing recognition of the importance of a catchment-based approach to land management, WEPP may not be adequate. WEPP may conceivably be extended to a larger scale by using a grid-based approach, that is, a series of hillslopes, however, this is likely to lead to problems associated with cumulative error. Thirdly, the WEPP model does not account for gully erosion or erosion from continuously flowing streams and thus may underestimate the impact a land use will have on erosion. In many Australian river systems, such as the Murrumbidgee, in-stream processes and gully erosion are the largest contributors to sediment load, yet these are ignored within the WEPP model. Finally, the rill–interrill concept of erosion used by WEPP may not be applicable in soils that have not been cultivated and do not initially exhibit rill formations.

The application of WEPP within Australia has been very limited, due to the complex model code and large parameter requirements. Fogarty (1997) carried out some initial testing of the model, concluding that WEPP could reliably predict runoff from disturbed forest land, and had the potential to predict sediment yield from the land. WEPP could potentially serve a useful role in predicting sediment yields, principally from disturbed forest land. However, there is very little literature, apart from the above, from Australia. Most likely this reflects the large computational requirements and inadequacy of the WEPP model within much of Australia due to the highly recognised lack of data.

Realistically, it would seem that the main use of WEPP within Australia will be limited to specific applications where there are sufficient data and funds to run the model.

Examples of Model Users:
Department of Land and Water Conservation, Queanbeyan.

Hardware Requirements:
PC under DOS operating system—at least 80386 CPU with a maths coprocessor. At least 10 MB free space on hard drive (more depending on simulations).

Availability/Cost:
Partial installation of v98.4 at ftp://soils.con.pursue.edu/pub/wepp/weppnpg.984. Beta version of WEPP Windows 95/NT Interface available from http://topsoil.nserl.purdue.edu/weppmain/wpslp.html.

4 IMPLEMENTATION OF MODELLING APPROACHES

Implementation of different modelling approaches in specific situations relies on a number of factors. Firstly the quality of fit of models, in particular conceptual and physics-based models, depends on the calibration acceptance criteria used by the modeller. The choice of model and its suitability to different tasks will also affect the quality of model fit and the usefulness of model implementation. This section provides an overview of calibration acceptance criteria, as well as a discussion of the factors affecting the predictive capacity of models, such as model complexity and modelling objectives.

4.1 Calibration acceptance criteria

Where calibration of model parameters against observed data is necessary, such as for conceptual and physics-based models and for some empirical models, two main types of approaches may be employed. The first is a subjective approach, relying on the modeller using a trial and error approach to estimate parameters. Certain statistics of fit are used to choose between sets of model parameters. This process is completed when the parameters are sufficiently accurate for the purposes of the model, according to the calibration criteria being used, not necessarily when the parameters are optimal. The second method is an objective approach, applying an algorithm to optimise parameter values using some single measure of goodness of fit, which may be a composite of several measures. As noted by Sorooshian and Gupta (1983), the main difficulty in this approach is that of finding a global optimum, as the non-convexity of the response surface may lead to the existence of a number of local optima. Generally parameter values are not unique (identifiable) and may not be physically realistic. This can lead to poor predictive ability on data periods independent of the calibration period in conceptual and physics-type models. The calibration process introduces an empirical element to the models, and limits the physical relevance of model parameters.

Generally model calibration is controlled by the use of one or more measures of 'goodness of fit' of modelled values to observed streamflow discharge and/or nutrient/sediment concentration or load data. It will not always be the case that these statistics are optimised for the same set of parameter values. An additional difficulty in finding optimal parameter sets is presented by the dependence of optima on the calibration criteria or objective function used for model calibration (Johnston and Pilgrim 1976). A parameter set that is optimal for one criterion is not necessarily, and indeed is unlikely, to be the same as the optimal parameter set using a different criterion. Calibration criteria are generally chosen subjectively. These need to be chosen to best suit the requirements of the model being calibrated. Questions on the intended use of the model and the nature of the catchment and problems being considered must be considered when choosing appropriate calibration criteria. Also some trade-off between the values of different statistics is usually necessary. Often it is best to optimise parameter values using a set of such measures rather than a single measure. When using such a set of measures it is not normally possible to optimise with respect to all measures simultaneously; rather, acceptable boundaries for each measure may need to be set.

Generally these calibration criteria are calculated on individual values of streamflow and/or sediment/nutrient discharge. However it is also possible to calculate these measures for different data sets, such as for hydrographs and sediment graphs of observed and modelled values or for different time periods, such as weeks and months, than used for modelling the original data series. The choice of which of these data sets to use will depend on the needs of the modeller and the characteristics of the data and catchment under consideration.

4.1.1 Mean and standard deviation

The most basic requirement of a model is generally that it describes the mean and standard deviation of observed data well. Thus a very simple measure of the fit of a model is to look at the agreement between the mean and the standard deviation of observed and modelled values. This simple measure does not distinguish between random and systematic errors and does not indicate how well individual estimated values fit observed values (Aitken 1973). Thus its usefulness in model calibration is limited when used alone.

4.1.2 Coefficient of Determination

The coefficient of determination is given by:

$$D = 1 - \frac{\sum_{i=1}^{n} \left(O_i - O_i^{est}\right)^2}{\sum_{i=1}^{n} \left(O_i - \overline{O}\right)^2} \tag{4.1}$$

where O_i are individual observed values, $\overline{O}$ is the mean of the observed values, P_i are individual modelled (or predicted) values and O_i^{est} is determined from regression of O_i on P_i.

The coefficient of determination measures the degree of association between observed and modelled values. It has value less than one for all models. High values of the coefficient indicate that the model is of good fit, however, by itself the coefficient is not able to reveal the presence of systematic errors.

4.1.3 Coefficient of Efficiency

The coefficient of efficiency (E) was described by Nash and Sutcliffe (1970) as

$$E = 1 - \frac{\sum_{i=1}^{N}(O_i - P_i)^2}{\sum_{i=1}^{N}(O_i - \overline{O})^2} \qquad (4.2)$$

This coefficient is analogous but not identical to the coefficient of determination. It describes the degree of association between the observed and modelled values of the data series. As for the coefficient of determination, values of E are less than or equal to1. The coefficients of determination and efficiency can be used together to determine whether model results are biased. If the model results are highly correlated but biased, then the value of E is less than that of D.

4.1.4 Least squares criteria

One common method for model calibration is to minimise the sum of squares of model error; that is, to choose model parameters such that

$$L = \sum_{i=1}^{n}\left(O_i - P_i\right)^2 \qquad (4.3)$$

is a minimum. The use of such a criteria to optimise model fit can be based on the assumptions that these errors are uncorrelated and that they have constant variance with zero mean. These assumptions are generally not satisfied.

This measure is equivalent to maximising the value of the coefficient of efficiency.

4.1.5 Absolute mean deviations

A more general technique is to minimise

$$A = \sum_{i=1}^{n}\left|O_i - P_i\right|^j \qquad (4.4)$$

where j is some exponent. The case where j = 2 is the least squares criterion of 4.1.4.

Changing the value of the exponent j merely changes the vertical scaling of the optimisation space, not the position of the minimum point (Johnston and Pilgrim 1976). However, values of j smaller than 1 generally make it difficult to optimise the parameter set as the observation space becomes flatter and more discontinuities are generated. Johnston and Pilgrim (1976) suggest that generally j = 2; that is, the least squares criterion described in 4.1.4 is the best value of j for optimisation.

4.1.6 Transformed deviations

It may be preferable in some cases to minimise deviations of transformations of the original and modelled values, using an objective function of the form

$$T = \sum_{i=1}^{n} \left| f(O_i) - f(P_i) \right|^j \tag{4.5}$$

where f is a function transforming observed and modelled values. Chiew and McMahon (1994) use the function $f(x) = x^{0.5}$. This provides weighting to reflect the performance of the model in simulating low flows. In an earlier paper Chiew *et al.* (1993) used a similar function, $f(x) = x^{0.2}$, for evaluating hydrological models. Generally when using a power transformation of the form $f(x) = x^a$ $a > 0$, the smaller the value of a, the more weight is given to model performance on low flows. Thus this calibration criterion may be most useful where the fit to low flows is at least as important as the fit of the model to peak flow events, such as in ephemeral catchments or in situations where low flows may determine important ecological characteristics of the catchment being modelled.

Other transformations commonly used include log transform functions.

4.1.7 Model bias

Model bias (B) is given by:

$$B = \frac{1}{n} \sum_{i=1}^{n} (O_i - P_i). \tag{4.6}$$

The model bias measures the average difference between the model outcome and the observed data. The magnitude of the bias must be near minimum to improve the model fit; that is, the closer the model bias is to 0 the better the model fits the observed data.

A similar measure to the model bias is the deviation in volumes (DV), given by:

$$DV = \frac{\sum_{i=1}^{n} P_i}{\sum_{i=1}^{n} O_i}. \tag{4.7}$$

The closer the value of DV to 1, the better is the model fit. Minimising model bias is equivalent to maximising the value of DV.

These measures are most useful when the modeller is mainly concerned with approximating the total volume of flow or pollutant over longer time periods, rather than with closely fitting the model to each observed value. This may be the case where model outputs are to be used to calculate monthly or yearly volumes of flow or pollutants, rather than for estimating daily or hourly flows of water or pollutants as a result of a rainfall event.

4.1.8 Serial correlation coefficient

A method occasionally used by hydrologists to indicate the presence of systematic errors is to compare the first order (and other orders) of serial correlation coefficient for observed and modelled values. It is doubtful whether this test is significantly powerful for this purpose (Aitken 1973). Thus this measure is probably best avoided when selecting calibration criteria.

4.1.9 Sign tests

Sign tests are not widely used by hydrologists. They are very simple tests of whether a modelled time series contains systematic errors. One possible sign test is to allocate a positive sign to overestimated values, and negative signs to underestimated values, then to count the number of runs of positive and negative signs and compare this with the expected numbers using a Chi Square test[1]. If this test indicates that the number of runs is significantly less than expected for random errors then it can be concluded that the model introduces systematic bias.

Such simple sign tests are the most suitable to quickly determine the existence of systematic errors. Aitken (1973) suggests that, as an initial step in testing for systematic errors, these tests should always be used in addition to more commonly used statistics when calibrating a model.

4.1.10 Maximum range of the residual mass curve

The residual mass curve for both modelled and observed values can be calculated by subtracting the mean value from each individual value, then summing the results sequentially. These curves for observed and modelled values can then be compared using such measures as the percentage error in the maximum range of the modelled residual mass curve. This method is not currently widely used within hydrology.

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[1] A chi square test consists of constructing a test statistic of the form

$$X = \sum_{j=1}^{k} \frac{\left(o_j - e_j\right)^2}{e_j}$$

where o_j is the observed number of runs of length j, and e_j is the expected number of runs of length j; then comparing this with the chi square distribution to determine whether the number of runs is less than expected for random errors, i.e. the value of X is too large.

4.1.11 Residual mass curve coefficient

The residual mass curve coefficient describes the association between the observed and modelled residual mass curves. It is given by:

$$R = 1 - \frac{\sum_{i=1}^{n}\left(d_i - d_i^e\right)^2}{\sum_{i=1}^{n}\left(d_i - d\right)^2} \tag{4.8}$$

where d_i is the departure from the mean for the observed residual mass curve, d is the mean of departure from the mean of the observed residual mass curve and d_i^e is the departure from the mean of the modelled residual mass curve.

The residual mass curve coefficient is better than the coefficients of determination or efficiency in describing model fit, as it measures the relationship between sequences of values, not simply between individual values. This coefficient should also indicate the presence of systematic errors. However, this coefficient is not currently widely used within hydrology, unlike the coefficient of efficiency (Aitken 1973).

4.1.12 Average relative parameter error

The average relative parameter error is given by

$$\text{ARPE} = \frac{1}{n}\sum_{i=1}^{n}\frac{\hat{\sigma}_i^2}{a_i^2} \tag{4.9}$$

where $\hat{\sigma}_i^2$ is the estimated variance of the ith element in the n parameter set $(a_1, a_2 \,..\, , a_n)$.

The ARPE is a measure of the average relative error in the model parameters. When calibrating a model the goal is to achieve the lowest possible magnitude of ARPE; that is, the value of the ARPE must be as close to 0 as possible. This measure is generally used to identify the number of parameters that are appropriate in a model. A high value of the ARPE indicates a high degree of uncertainty in parameter estimates, which implies a poorly defined model.

4.1.13 Summary

The most commonly used of the calibration criteria described is the coefficient of efficiency. In practice, almost all model calibrations depend on this or a least squares criterion. Many of the other criteria described here have not been used within hydrology, or have been used only for modelling specific catchments, where particular factors such as model fit to low flows or model fit to total yearly observations are important.

Generally it is best to use a combination of several criteria that best fit the particular modelling situation.

4.2 Predictive capacity

A wide range of models exist for use in sediment transport and water quality modelling. These models differ in terms of complexity, the nutrients and processes considered and the data required for model use. There is no 'best' model; rather, the most appropriate model will depend on the intended use and the characteristics of the catchment being considered. A number of additional factors need to be considered in order to choose the appropriate model for an application. These can be summarised as the suitability of the model to Australian conditions, the ease of use and data requirements, the hardware requirements, the accuracy and validity of the model, the model assumptions, the spatial and temporal variation of model inputs and outputs, the components of the model, and the objectives of the model users, including the scale at which model outputs are required.

Table 4.1 shows a general description for many of the sediment transport and water quality models available for use. Of the models in the table, AEAM, AQUALM, CMSS, HYDRA, IHACRES, Rose and Hairsine Approach, SOILOSS, STARS, THALES and TOPOG are Australian-developed or -adapted models. Many of the other models were developed in the USA or Europe and have been used in Australia. Most of these models were developed by excluding or including processes appropriate to certain environments. Some, like EUROSEM/ LISEM, have had limited applications under Australian conditions and consequently would require a period of extended testing, validation and, if required, calibration to ensure that they are capable of accurately modelling Australian conditions.

Classification of models as empirical, conceptual or physics-based is subjective. Most models do not fit neatly into these categories; rather, they are likely to contain a mix of modules from each of these categories. For example, whilst the rainfall-runoff component of a model may be physics-based or conceptual, empirical relationships may be used to model erosion or sediment transport. The classification given for each model in Table 4.1 reflects the main processes in the model and does not mean that there are not components of the model better classified in another category. Where the mix of modules is fairly even, a model is classified as a hybrid between two or more classes.

4.2.1 Model complexity and ease of use

Ease of use is of considerable importance when choosing an appropriate model, the importance of which is driven largely by the objectives and capabilities of the model user. Rizzoli and Young (1997) identified two main categories of users with respect to user requirements: the 'scientist' (also known as the modeller or systems analyst) and the 'manager' (otherwise referred to as the decision-maker). With the development of Landcare and community-based catchment groups there is a trend towards the use of simple decision support systems as tools for establishing appropriate management practices. Subsequently, there has been an increase in the development and use of models such as CMSS, HYDRA and AEAM. These models, often termed Environmental Decision Support Systems or EDSSs, can be used to solve problems relating to a specific domain of knowledge, such as water quality in streams (Rizzoli and Young 1997). Often these models do not attempt to describe the physical processes involved. Rather, they rely on the use of simple, empirically determined relationships. The outputs of such models are often used as a basis for developing catchment management plans.

Such models tend not to require large quantities of data and are computationally simple. In contrast, the physics-based models such as WEPP, ANSWERS, and MIKE-11 require a large amount of input data and consequently can be difficult to use. This can be a particular problem in Australian catchments where input data is typically sparse. A large number of parameters in these models will have to be determined through calibration in such sparse data situations, raising difficulties with identifiability, model uniqueness, physical interpretability of calibrated parameters, and user friendliness. Conceptual models require calibration against observed data. They are also likely to suffer from problems of non-uniqueness and model identifiability. They too are most appropriate for use by an experienced modeller (Hook 1997). Some conceptual models have a small number of parameters making them more easily identifiable. The IHACRES rainfall-runoff model contains only six parameters and has been shown to work well across different climates and catchment sizes. While the LASCAM rainfall-runoff model contains around as many as 20 parameters, its sediment component contains only 6 parameters.

In addition, many conceptual and physics-based model users aim to incorporate components of other models into their own to tailor the model to their requirements. Some users of the WEPP model have found it very difficult to integrate its components with other models due to the complex structure of WEPP and the difficulty in penetrating the model code. This indicates another potential problem with complex conceptual and physics-based models. Generally, these models tend to be used for research or by experienced model users while the simple empirical or conceptual models or EDSSs are used by managers with limited data or modelling experience, or those who require flexibility in the modelling process.

4.2.2 Hardware requirements

Also of relevance to the model user is the hardware requirements of a model. This is determined by the complexity of the model, the processes that are represented in the model and the extent to which these processes are considered. Physics-based models, being based on the solution of fundamental physics equations, often require numerical solutions that have greater hardware requirements than empirically based models like USLE, which are computationally relatively simple and require little in the way of technically advanced hardware. In addition, many of the research models available use hardware and platforms (e.g. UNIX) not widely used by non-research groups. To make these models accessible either a modified version of the model or a Windows interface is required.

The hardware requirements are also determined by the detail of the catchment processes simulated. Not only do the number of equations requiring solution increase in a model with a large number of detailed processes, but so do the number of input parameters. For example, the detachment of soil particles, either through raindrop impact or the flow of water across the soil surface, is affected by a number of factors, including soil properties, topographical features and land use or land management practices, particularly those that influence the vegetation cover in the catchment. This has a profound impact on the modelling of sediment transport and water quality. An attempt to quantify the effect of all the parameters that affect sediment yield will result in a computationally exhausting process. Physics-based models, such as WEPP and ANSWERS, and many conceptual models require a large amount of input data, much of which is unavailable in most Australian catchments.

4.2.3 Accuracy and validity of model predictions

An important consideration in choosing models is the accuracy and validity of the model. This relates back to the issue of the suitability of a model to a particular environment. For example, in the USA, more than 10 000 plot years of data has been collected and incorporated into the USLE erosion models, such that it is applicable at the plot scale to those environments (Lane *et al.* 1992). In Australia, by contrast, the use of USLE has been limited due to the perceived lack of data. As such, the validity of the model under Australian conditions has been questionable. SOILOSS, a local adaptation of parts of a revised USLE model (RUSLE), attempts to address this problem to some extent. Similarly, PERFECT was developed for application in environments typical of north-eastern Australia. Nonetheless, Littleboy *et al.* (1992) recommended that the model be calibrated against suitable field data before being used in other environments.

Another common misconception is that model accuracy invariably increases with model complexity. This is not the case. Complex models such as WEPP suffer from problems with error accumulation and model identifiability. The lack of available input data for such models means that many of the model parameters must be determined through calibration. This leads to problems of non-uniqueness and means that the physical interpretability of parameter values is questionable. Additional errors may come from the use of unrealistic assumptions about the physics controlling the catchment system. For example, the MIKE-11 model is based on the solution of one-dimensional equations of flow. However, these equations are being used to represent a three-dimensional physical system. The accuracy of a model based on such unrealistic assumptions is questionable.

4.2.4 Model assumptions

The accuracy of any model will be determined in part by the assumptions underlying the model. For example, the USLE and WEPP hillslope and small-catchment models have been designed to study erosion in situations of overland flow. Thus they are intended to be used to model sheet and rill erosion, and are not suitable for situations where erosion is by channels that cannot be removed by tillage, such as gullies or streams. In situations where significant amounts of erosion occur by gully or stream, application of such models is likely to lead to large inaccuracies in model simulations. The use of a model that considers only rill and sheet erosion, such as WEPP, is likely to lead to a large underestimation of sediment and nutrient loads in areas where gully and in-stream erosion processes are important, such as in the Murrumbidgee River catchment. Such models would be inappropriate as they do not consider the main source of sediment and nutrients in such catchments. Similarly, the field-scale CREAMS and EPIC models assume that the site being modelled is uniform in soil, topography and land use. Application of these models at scales over which these characteristics are heterogeneous may lead to substantial errors. Finally, ANSWERS assumes erodibility to be a relatively time-constant parameter, contrary to the large variations that have been recorded (Govers and Loch 1993).

Hairsine and Rose (1992) noted that in past literature, soil erosion processes occurring during overland flow were considered to be very similar to those occurring during streambed erosion, and subsequently sediment transport equations derived for deep flow conditions have been used to describe the movement of sediment in the shallow flows characteristic of soil erosion on the field scale. However, the authors identified differences between the two erosion types in terms of the sedimentary material and the processes at work. At the field-scale, sediment is usually cohesive, having both interaggregate and interparticle strength; soils are commonly composed of a wide range of aggregate and particle sizes; and shallow surface flows that occur at field scales are influenced by the impact of raindrops on both the shallow water layer and the exposed soil surface (Hairsine and Rose 1992). Given these differences, it is not necessarily appropriate to use stream-based models for the prediction of overland flow erosion.

These examples show the types of assumptions common in environmental modelling. These assumptions have been made to simplify the model, but the modeller needs to keep them in mind, as they are likely to affect the accuracy of the model. Likewise, the simplification of in-stream processes in models like MIKE-11, such as the neglect of secondary currents and bank erosion processes, are likely to reduce the accuracy of the model. As previously stated MIKE-11 uses a one-dimensional approach to represent three-dimensional processes. Physics-based models such as WEPP and MIKE-11 tend to be based on equations that have been derived in laboratory conditions. These equations may not be applicable in real-world situations, where many of the initial conditions are likely to be different and a number of the assumptions are likely to be violated.

4.2.5 Topographic effects and spatial and temporal variability

The previous section indicates the importance of identifying the key hydrologic and erosion components in water quality modelling. There are, however, a number of other factors that need to be accounted for prior to choosing a model. These include the problems associated with spatial and temporal variability, topographic effects and the suitability of the model to the study objective of the site in question.

Jakeman *et al.* (1997) noted that the difficulties in environmental modelling can be characterised as problems of natural complexity, spatial heterogeneity and the lack of available data. The complexity of natural systems is due to the differences in dimensions, temporal and spatial scales and thresholds of water flow, and sediment and nutrient transport through and within the media. Natural systems, from plot to catchment scale, tend to show a great deal of variation. Grayson and Moore (1993) noted that the scale at which uniformity is assumed in hydrologic models is generally greater than the scale at which directly measurable parameters are measured in the field, although it is smaller than shown by the outflow hydrographs. Thus, model predictions are subject to errors as a result of the inconsistency of scale between measured parameters and the way they are used in the model.

Moore *et al.* (1991) suggested that a deficiency in many hydrologic and water quality models is the lack of representation of the effects of three-dimensional terrain on flow process and spatial variability of hydrologic processes with large and often unrealistic simplifications. Topographical features can potentially have a large effect on hydrologic and erosion processes and as such are an important consideration in water quality modelling. With the development of Digital Elevation Models (DEMs) and GIS, topographical attributes can be, and are increasingly, incorporated into water quality or hydrologic models (e.g. LISEM, THALES).

Many physics-based and conceptual models attempt to account for topographic effects. However, these models are often highly complex, containing large numbers of parameters often varying both spatially and temporally, and thus are used more by research organisations rather than by government departments or community groups.

4.2.6 Model components

Many models are designed to target a particular component of an environmental problem, such as the erosional, hydrologic or water quality component. For example, both USLE and WEPP are erosion models, and like THALES, a hydrologic model, fail to account for the ‘whole picture’ of erosion issues. Depending on the structure of the model, additional components, whether from other models or not, may be incorporated to further validate the model predictions. It should be noted that this may add to the complexity of the model. The PERFECT productivity–erosion model is an example of the flexibility of a model structure which can allow the incorporation of additional modules. The model, although currently used more as a model for identifying the effects of crop management on erosion and yield, may be able to incorporate a water quality component that could predict the impact of crop management practices on water quality.

Recently there has been an increase in the number of models developed for water quality and pollution issues (e.g. AGNPS, ANSWERS, AQUALM, HSPF, CMSS, MIKE-11). These models tend to incorporate a number of modules covering the hydrological, erosional and other components affecting water quality. These models tend to incorporate other models that are specifically designed for one purpose. For example, models like AGNPS and EPIC incorporate USLE-based erosional modules into the overall model structure. Likewise, PERFECT uses components from both USLE and CREAMS for erosion and sediment transport.

4.2.7 Objectives of the model user

Finally, the objectives of the modeller will perhaps be the largest factor influencing the choice of model. This will largely determine complexity and depth of understanding of the model structure and purposes required. Model types can be broadly categorised into empirical, conceptual and physics based models. Physics-based models tend to be complex models, aimed at furthering knowledge of some of the processes involved in sediment and nutrient generation. These models (e.g. WEPP and MIKE-11) tend to be used more by researchers for detailed projects.

Simpler empirical or conceptual models are not specifically aimed at fully understanding the processes involved in sediment and nutrient generation. The complexity of the model will determine how the model is used. For example, a common approach in water quality catchment management programs is to use empirical models, often referred to as Environmental Decision Support Systems (EDSSs), such as CMSS and AEAM. CMSS is based on a simple nutrient load model using empirical relationships between nutrient generation and land use. It is generally used to assist land use and land management planning for water quality improvement at catchment scale. The model has been used by 60 registered users throughout Australia and New Zealand by state and local government agencies, Total Catchment Management or Integrated Catchment Management groups and consultancy firms (Hook 1997). This illustrates the importance of these simpler management models that do not give a definitive solution to a problem but allow the model user to develop best management practices for the site. The most appropriate model for a given situation will therefore depend on whether the aim of the modeller is to accurately predict catchment yields in gauged or ungauged catchments, to better understand the processes generating sediment and nutrients in the catchment, or to assess the likely impacts of a change in catchment management. The best model will depend on the resources available to the modeller as well as the required accuracy and outputs of model simulation.

Thus the choice of the model most appropriate in any situation is dependent on a number of factors unique to the modelling situation. Consideration needs to be given to the requirements of the situation and the resources, including the input data, computing resources and modelling expertise, available. Section 3.2 gives a concise description of a number of erosion and sediment/nutrient transport models available. It also provides a description of model inputs and outputs, a discussion of model limitations and advantages, and information on hardware requirements and availability for each model.

4.2.8 Synthesis

The data sets on sediment and nutrient concentrations are typically available only at large catchment scales of the order of 100 to 1000 km^2, as well as for a limited temporal period, often only up to a few years. Such information is inadequate to support the application of complex models which contain large numbers of parameters and/or which make detailed assumptions about the physical processes driving transport. Only the key catchment processes warrant description in such data-poor circumstances.

Complex conceptual and physics-based models also place high demands on the user, who must be very experienced technically in using models. Even for the experienced, the unique calibration of so many parameters is not possible. Different users will therefore obtain different parameter sets.

In addition, physics based models are typically designed to be applied only at small scales. Their application to larger scales brings attendant problems of high computational requirements and error accumulation.

Therefore, it is only empirical models and simple conceptual models which can be considered as suitable for modelling catchment exports at catchment and basin scales.

Conceptual models typically have two components, one for routing rainfall-runoff processes and one linking the routing of water to solute concentration. Such models include LASCAM and HSPF, but both these models contain unnecessarily complex descriptions of the rainfall-runoff process. LASCAM does, however, contain a runoff-sediment component of reasonably low complexity (only six parameters). The model IHACRES contains a parsimonious rainfall-runoff description of six parameters, which has been shown to work well in hundreds of catchments in predicting streamflow discharge across a range of scales and hydroclimates. To predict catchment exports it has been augmented with empirical models (e.g. power law relations) of discharge and suspended sediment, and suspended sediment and nutrient concentrations. The conceptual runoff-sediment component of LASCAM could also be augmented with IHACRES to provide a conceptual model of catchment exports with reasonable complexity.

However, any conceptual modelling approach needs to take into account the fact that, in many Australian catchments, streambank erosion is a major source of sediments and phosphorus. The STARS model was specifically developed to model this process, allowing the identification of sources (bank erosion, tributary inflows) and sinks (bed deposition) within river reaches. Its structure was designed to be simple, containing only five parameters, so that it could be calibrated successfully on time series of upstream and downstream discharge and pollutant concentrations.

Of the empirical approaches, direct load estimation techniques must be considered seriously. These are particularly suitable if the observations available for estimation span a climatic range covering wet and dry periods so that loads can be calculated as long-term values accompanied by a measure of their variability. As discussed in Chapter 2, the appropriate load estimation technique will vary with the nature of the data and the catchment conditions.

Another type of empirical approach that is useful for its potential in being applied to ungauged catchments or subcatchments is that based on land use/landscape attributes. Here the classification of land use must be sufficiently broad so that the export results are sensitive to members of the classification (such as pristine, cropping, grazing). These models, such as the multi-factor approach of Moss *et al.* (1993) or approaches embodied in CMSS, have the disadvantage that the export values produced are not directly sensitive to climate variability (of events and antecedent conditions). They tend to yield long-term averages only. However, it is possible for these multi-factor models to be integrated with either direct load estimation techniques or simple conceptual models so that the resultant export outputs are climate-sensitive.

In conclusion, given the problems with complex conceptual and physics-based models—i.e. those of model parameter identifiability, computational demands and necessary levels of user expertise—the most practicable approach is one integrating the use of direct load estimation, multi-factor and simple conceptual models. Direct load estimation will work best when predicting exports at sites which have lengthy but intensive data, and will provide information to calibrate and/or corroborate the other two model types. Multi-factor models will be useful when predicting at ungauged sites, such as when direct load estimation or conceptual models require predictions at subcatchment or even landscape scales where no measurements are available. That is, they will be useful for disaggregating the exports predicted at larger catchment scales and lend themselves to being incorporated in conceptual models as subcatchment-scale predictions. Conceptual models will be useful especially to link subcatchment exports and route them through catchment and basin networks. As runoff and discharge are the major drivers of catchment exports, a good conceptual model will be one which:

- predicts runoff from catchments and routes discharge and pollutants through an instream component
- incorporates the key processes (quick flow, slow flow, stream advection, suspension and resettling) in a parametrically efficient manner. With climate being the major determinant of long-term variability of catchment exports, a conceptual model, which allows forcing from rainfall and other climate variables (such as temperature), is essential to help characterise the variability of exports.

Table 4.1: Erosion/Sediment Transport Models

Model	Type	Scale	Output	Event/ Non-event	Part or complete GIS integration	Spatially distributed	Comments
AEAM	Empirical/ Conceptual	catchment	depends on model application or user requirements	non-event	depends on particular model	no	Modelling process rather than set model structure. Environmental Decision Support System. Similar to CMSS, although slightly more complex equations
AGNPS	Conceptual	catchment (up to 50 000 ha)	runoff volume, peak rate, eroded and delivered sediment, N, P and COD concentrations in runoff and sediment	event	yes (can be linked)	yes	One of few models whose output is site- and management-specific
ANSWERS	Physical	< 4000 ha	sediment yield, nutrient loads in water and sediment runoff	event	yes	yes	
AQUALM	Conceptual	catchment	nutrient load	event	yes	–	In-stream model, similar to HSPF with simpler algorithms and explains fewer processes
CMSS	Empirical	catchment	nutrient load	non-event	no (though land use attributes may be obtained by GIS)	yes	Aimed at catchment managers, community awareness
CREAMS	Physical	field size 40–400 ha	erosion, deposition, transport (slope to 2nd-order channels)	event	no	no	Catchment assumed to be uniform in soils, topography and land use
EPIC	Conceptual	field size, < 100 ha	nutrients, sediments, runoff, pesticides, plant growth	event	yes	–	Weather, soils and management considered homogenous; considers N, P, pesticides, sediment; USLE based

Model	Type	Scale	Output	Event/ Non event	Part or complete GIS integration	Spatially distributed	Comments
EUROSEM/ LISEM	Physical	catchment	runoff, sediment yield	event	LISEM linked to PCRaster (GIS)	yes	Does not model erosion in rills and gullies
GUEST	Physical	field	soil loss predictions	event	–	–	Uses the Rose and Hairsine approach (1997)
HSPF	Conceptual	catchment	runoff, flow rate, sediment load, nutrient concentration, water quality	both (different versions)	yes	yes	Developed from Stanford Watershed model. Relies heavily on calibration against field data
HYDRA	Empirical/ conceptual	catchment	water quality management	non-event	uses GIS based interface	–	Aimed to integrate current modelling practices into an Environmental Decision Support System
IQQM	Conceptual	river basin	BOD, coliforms, nitrogen cycle, phosphorus cycle, flow routing	non-event	–	–	
MIKE-11	Physical	catchment	sediment yield, runoff	event	–	yes	One-dimensional water quality model
PERFECT	Physical	field	runoff, erosion, crop yield	event	–	–	Incorporates a crop growth simulation module
Modified LASCAM	Conceptual	catchment	rainfall runoff, salt fluxes, sediment	event	no	can be	Modified by inclusion of USLE component
Rose and Hairsine Approach	Physical	small scale	sediment yield	event	–	–	Problems occur when there is an attempt to parameterise over large scales; model implemented in other models e.g. GUEPS.
SOILOSS	Empirical/ conceptual	hillslope	average annual soil loss	non-event	yes	no	Australian (NSW) adaptation of RUSLE

Model	Type	Scale	Output	Event/ Non event	Part or complete GIS integration	Spatially distributed	Comments
STARS-IHACRES	Conceptual	catchment	runoff, sediment, nutrient concentrations	event	can be integrated	can be	IHACRES is the upland runoff component and STARS is the instream water quality component
SWMM	Physical	catchment	overall assessment of urban runoff, prediction of flows, stages and pollutants	both	can be integrated in similar way to AGNPS	yes	Rainfall runoff simulation model, primarily for urban areas (i.e. point source)
THALES	Physical	catchment	wide range of hydrological processes	–	yes	yes	Using TAPES-C computer program, accounts for 3D terrain. Assumptions underlying THALES are extensive
TOPOG	Physical	catchment	water, solute, carbon balance	event	Uses DEM	yes	Three components: DEM, topographical analysis model, suite of hydrological and process models. Model based on spatial analysis and mathematical models of a wetness index and stream power
USLE/RUSLE/ MUSLE/ USLE-M	Empirical/ conceptual	hillslope	average annual soil loss due to rainfall	non-event (modified versions may be event based)	yes	no (can model spatial variation when considered in grid)	Many modifications of the original model (MUSLE, USLE-M). Model was revised to include new information (RUSLE). This revised USLE was implemented locally in the SOILOSS model. Does not model gully or in-stream erosion
WEPP hillslope model	Physical	hillslope	runoff, sediment, form of sediment loss	both	yes	yes	Does not account for gully erosion or mass movement
WEPP watershed model	Physical	hillslope	runoff, sediment, form of sediment loss	both	yes	yes	Watershed model comprises hillslope model with channel erosion component, impoundment component, and irrigation component

REFERENCES

Chapter 1

1. Attiwill, P.M., and Leaper, G.W. (1987). 'Forest soils and nutrient cycles.' Melbourne University Press, Carlton, Victoria.

2. Bennett, J. P. (1974). 'Concepts of mathematical modelling of sediment yield.' *Water Resources Research*, **10**, 485-492.

3. Brizga, S.O., and Finlayson, B.L. (1990). 'Channel avulsion: its role in river metamorphosis.' *Earth Surface Processes and Landforms* **15**, 391-404.

4. Conner, T.C. (1986). 'The impact and economic effects of soil conservation practices on river aggregation.' In Hadley, T.F. (ed.). *Drainage basin sediment delivery*' IAHS publication No. 159. IAHS Press, Wallingford, U.K. pp. 81-91.

5. Correll, D.L. and Ford, D. (1982). 'Comparison of precipitation and land runoff as sources of estuarine N.' *Estuarine Coastal Shelf Science*, **15(1)**, 45-56.

6. Cosser, P. R. (1989). 'Nutrient concentration – flow relationships and loads in the South Pine River, South-Eastern Queensland. 1. Phosphorus loads.' *Australian Journal of Marine and Fresh Water Research,* **40**, 613-630.

7. Eyre, B. (1997). 'Water quality changes in a well flushed Australian estuary: A 50 year perspective.' *Marine Chemistry,* **59**, 177-187.

8. Finlayson, B., and Silburn, M. (1996). 'Soil, nutrient and pesticide movements from different land use practices, and subsequent transport by rivers and streams.' In Hunter, H.M., Eyles, A.G., and Rayment, G.E. (eds.). *Downstream Effects of Land Use.* Department of Natural Resources, Queensland, Australia.

9. Frissel, M. J., (ed.). (1978). *Cycling of Mineral Nutrients in Agricultural Ecosystems.*, Elsevier Scientific, Amsterdam.

10. Gerritse, R.G. (1995). 'Effect of reaction-rate on leaching of phosphate through sandy soils in Western Australia', *Australian Journal of Soil Research*, **33(1)**, 211-219.

11. Grayson, Gippel, C. J., Finlayson, B. L., Hart, T. B., Hawken, R., McMahon, T.A., and Tilleard, J. (1994). Mitigation of sediment sources in the Latrobe River catchment into the Gippsland Lakes. Completion Report on LWRRDC Project UME8. Centre for Environmental Applied Hydrology, University of Melbourne.

12. Grayson, R. B., Finlayson, B. L., Gippel, C. J., and Hart, T. B. (1996). 'The potential of field turbidity measurements for the computation of total phosphorus and suspended solids loads.' *Journal of Environmental Management,* **47**, 257-267.

13. Hinga, K.R., Keller, A.A., and Oviatt, C.A. (1991). 'Atmospheric deposition and nitrogen inputs to coastal waters.' *Ambio*, **20(6)**, 256-260.

14. Hossain, S. (1998). Hydrology and suspended sediment transport in the Richmond River catchment and estuary, NSW, Australia. PhD thesis, Southern Cross University.

15. Lane, S. (1998). 'Hydraulic modelling in hydrology and geomorphology: a review of high resolution approaches.' *Hydrological Processes*, **12(8)**, 1131-1150.

16. McKee, L. J., and Eyre, B. D. (1996). 'Nutrient export in a large rurally diverse northern NSW coastal catchment: A preliminary assessment.' In Hunter, H.M., Eyles, A.G., and Rayment, G.E. (eds.). *Downstream Effects of Land Use.* Department of Natural Resources, Queensland, Australia.

17. Meade, R.H. (1982). 'Sources, sinks and storage of river sediments.' *Journal of Geology,* **90**, 235-252.

18. Meybeck, M. (1982). 'Carbon, nitrogen, and phosphorus transport by world rivers.' *American Journal of Science*, **282**, 401-450.

19. Moss, A. J., Rayment, G. E., Reilly, N., and Best, E. K. (1993). A preliminary assessment of sediment and nutrient exports from Queensland coastal catchments. Environment Technical Report No. 5. Queensland Department of Environment and Heritage and Queensland Department of Primary Industries, Brisbane.

20. Novotny, V., and Chesters, G. (1989). 'Delivery of sediment and pollutants from nonpoint sources: A water quality perspective.' *Journal of Soil and Water Conservation,* **44**, 471-478.

21. Rutherford, I.D., and Smith, N. (1992). Sediment sources and sinks in the catchment of the Avoca River, north western Victoria. Report No. 83, Water Division, Department of Conservation and Natural Resources, Melbourne.

22. Sharpley, A. N., and Menzel, R. G. (1987). 'The impact of soil and fertiliser phosphorus on the environment.' *Advances in Agronomy,* **41**, 297-324.

23. Simeoni, M, Hickey, C., Gillespie, L., Kachka, A. and Vorreiter, L. (1994). Stormwater Monitoring Project 1993 Annual Report. Prepared for Clean Waterways Programme, AWT Ensight, Waterboard, Sydney, Report No. 94/93.

24. US Environmental Protection Agency (1983). Results of the Nationwide Urban Runoff Program Volume I. Final Report. USEPA Water Planning Division, Washington, DC.

25. Wasson, R. J., Donnelly, T. H., and Murray, A. S. (1996). 'Imports can be dangerous—Appropriate approaches to Australian rivers and catchments.' *First National Conference on Stream Management in Australia.*

26. Wasson, R. J., Mazari, R. K., Starr, B., and Clifton, G. (1998). 'The recent history of erosion and sedimentation on the Southern Tablelands of southeastern Australia: sediment flux dominated by channel incision.' *Geomorphology* **24**, 291-308.

27. Webb, B. W., and Walling, D. E. (1982). 'Magnitude and frequency characteristics of fluvial transport in a Devon drainage basin and some geomorphological implications.' *Catena*, **9**, 9-23.

28. Young, T. C., and DePinto, J. V. (1988). 'Factors effecting the efficiency of some estimators of fluvial total phosphorus load.' *Water Resources Research,* **24**, 1535-1540.

29. 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.

30. Zhang, L., O'Neill, A., and Lacy, S. (1995). 'Spatial analysis of soil erosion in catchments: a review of modelling approaches.' *International Congress on Modelling and Simulation (MODSIM95)*, Water Resources and Ecology, **3**, 58-64.

Chapter 2

1. Adams, J. (1980). 'High sediment yields from major rivers of the western Southern Alps, New Zealand.' *Nature,* **287**, 88-89.

2. Alberts, E. E., Schuman, G. E., and Burwell, R. E. (1978). 'Seasonal runoff losses of nitrogen and phosphorus from Missouri Valley loess watersheds.' *Journal of Environmental Quality,* **7**, 203-208.

3. ANZECC (1992). *Guidelines for protection of fresh and marine waters.* Australian and New Zealand Environment and Conservation Council.

4. Beaulac, M. N., and Reckhow, K. H. (1982). 'An examination of landuse – nutrient export relationships.' *Water Resources Bulletin,* **18**, 1013-1024.

5. Billen, G., Somville, M., De Becker, E., and Servais, P. (1985). 'A nitrogen budget of the Scheldt hydrographical basin.' *Netherlands Journal of Sea Research,* **19**, 223-230.

6. Birch, P. B. (1982). 'Phosphorus export from coastal plain drainage into Peel–Harvey estuarine system.' *Australian Journal of Freshwater Research,* **33**, 23-32.

7. Boynton, W. R. Garber, J. H. Summers, R., and Kemp, W. M. (1995). 'Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries.' *Estuaries,* **18**, 285-314.

8. Caraco, N. F. (1995). *Influence of human populations on P transfers to aquatic ecosystems: A regional study using large rivers.* Scope, John Wiley & Sons Ltd. pp. 235-244.

9. CSIRO (1992). *Towards Healthy Rivers*: A report from CSIRO to the Honourable Ros Kelly, Minister for Arts, Sport, the Environment and Territories. Wasson, R.J. (ed.). Division of Water Resources. Consultancy Report No. 92/44, November, 1992. Australian Government Publishing Service, Canberra.

10. Clarke, R. (1990). 'Bias and variance of some estimators of suspended sediment load.' *Hydrological Sciences Journal,* **35**, 253-261.

11. Cohn, T., Delong, L., and Gilroy E. (1989). 'Estimating constituent loads.' *Water Resources Research* **25**, 937-942.

12. Cole, J. J., Peierls, B. L., Caraco, N. F., and Pace, M. L. (1993). 'Nitrogen loading of rivers as a human driven process.' In *Humans as components of ecosystems: the ecology of subtle human effects and populated areas*. M. J. McDonnell and S. T. A. Pickett (eds.). New York, Springer-Verlag: 141-157.

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

14. Correll, D. L. (1981). 'Nutrient mass balances for the watershed, headwaters, intertidal zone, and basin of the Rhode River Estuary.' *Limnology and Oceanography,* **26**, 1142-1149.

15. Correll, D. l., Jorden, T. E., and Weller, D. E. (1992). 'Nutrient flux in a landscape: Effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters.' *Estuaries,* **15**, 431-442.

16. Cosser, P. R. (1989). 'Nutrient concentration – flow relationships and loads in the South Pine River, South-Eastern Queensland. 1. Phosphorus loads.' *Australian Journal of Marine and Fresh Water Research,* **40**, 613-630.

17. Cullen, P., Rosich, R., and Bek, P. (1978). *A Phosphorus Budget for Lake Burley Griffin and Management Implications for Urban Lakes*. Tech. Paper No. **31**, AWRC.

18. Davis, J. S., and Keller, H. M. (1983). ‘Dissolved loads in streams and rivers—discharge and seasonally related variations’. In Webb, B.W. (ed.). Dissolved loads of rivers and surface water quantity/quality relationships. *Proceedings of the Hamburg Symposium, August 1983.* International Association of Hydrological Sciences (IAHS).

19. Dillon, P. J., and Kirchner, W. B. (1975). ‘The effects of geology and land use on the export of phosphorus from watersheds.’ *Water Research,* **9**, 135-148.

20. Dorioz, J. M., Cassell, E. A., Orand, A., and Eisenmann, K. G. (1998). ‘Phosphorus storage, transport and export dynamics in the Foron River watershed.’ *Hydrological Processes,* **12**, 285-309.

21. Eyre, B., and McKee, L. (1998). *A nutrient budget for Moreton Bay, southeast Queensland, Australia (Task NB).* Report prepared for the Brisbane River and Moreton Bay Wastewater Management Study. Centre for Coastal Management, Southern Cross University.

22. Eyre, B., Hossain, S., and McKee, L. (1997). *Factors controlling turbidity in the Brisbane River estuary (Task ETP).* Report prepared for the Brisbane River and Moreton Bay Wastewater Management Study. Centre for Coastal Management, Southern Cross University.

23. Ferguson, R. I. (1986). ‘River loads underestimated by rating curves.’ *Water Resources Research,* **22**, 74-76.

24. Ferguson, R. I. (1987). ‘Accuracy and precision of methods for estimating river loads.’ *Earth Surface Processes and Landforms,* **12**, 95-104.

25. Finlayson, B., and Silburn, M. (1996). ‘Soil, nutrient and pesticide movements from different land use practices, and subsequent transport by rivers and streams.’ In Hunter, H.M., Eyles, A.G., and Rayment, G.E. (eds.). *Downstream Effects of Land Use*. Department of Natural Resources, Queensland, Australia.

26. Freifelder, R. R., Smith, S. V., and Bennett, R. H. (1998). ‘Cows, humans and hydrology in the nitrogen dynamics of a grazed rural watershed.’ *Journal of Environmental Management,* **52**, 99-111.

27. Frink, C. R. (1991) ‘Estimating nutrient exports to estuaries.’ *Journal of Environmental Quality,* **20**, 717-724.

28. Frissel, M. J., (ed.). (1978). *Cycling of mineral nutrients in agricultural ecosystems.* Amsterdam, Elsevier Scientific.

29. Gippel, C.J. (1989) ‘The use of turbidimeters in suspended sediment research.’ *Hydrobiologia*, **176**(Jul), 465-480.

30. Grayson, R. B., Finlayson, B. L., Gippel, C. J., and Hart, T. B. (1996). 'The potential of field turbidity measurements for the computation of total phosphorus and suspended solids loads.' *Journal of Environmental Management,* **47**, 257-267.

31. Griffiths, G. (1979). 'High sediment yields from major rivers of the western Southern Alps, New Zealand.' *Nature,* **282**, 61-63.

32. Groth, E., King, A. L., and Malone, C. R. (eds.) (1978). *Nitrates: An environmental assessment*. National Academy of Sciences, Washington, D.C.

33. Haith, D. A., and Shoemaker, L. L. (1987). 'Generalized watershed loading functions for stream flow nutrients.' *Water Resources Bulletin,* **23**, 471-478.

34. Heathwaite, A.L., Johnes, P.J. and Peters, N.E. (1996). 'Trends in nutrients'. *Hydrological Processes,* **10**, 263:293.

35. Hill, R. D., and Peart, M. R. (1998). 'Land use, runoff, erosion and their control: a review for southern China.' *Hydrological Processes,* **12**, 2029-2042.

36. Hoare, R. A. (1982). 'Nitrogen and phosphorus in the Ngongotaha Stream.' *New Zealand Journal of Marine and Freshwater Research,* **16**, 339-349.

37. Howarth, R. W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J. A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P., and Zhao-Liang, Z. (1996). 'Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences.' *Biogeochemistry,* **35**, 75-139.

38. Hoyas, T. R., Vagstad, N., Bechmann, M., and Eggestad, H. O. (1997). 'Nitrogen budget in the River Auli catchment: A catchment dominated by agriculture, in southeastern Norway.' *Ambio,* **26**, 289-295.

39. Hunter, H. M. (1993). Fluxes of nitrogen, phosphorus and suspended solids in the Johnstone river system: April 1991 – March 1993. Unpublished report ACU93.19 Department of Primary Industries, Brisbane.

40. Jaworski, N. A., Goffman, P. M., Keller, A. A., and Prager, J. C. (1992). 'A watershed nitrogen and phosphorus balance: The upper Potomac River Basin.' *Estuaries,* **15**, 83-95.

41. Jordan, T. E., Correll, D. L., and Weller, D. E. (1997). 'Nonpoint source discharges of nutrient from piedmont watersheds of Chesapeake Bay.' *Journal of the American Water Resources Association,* **33**, 631-645.

42. Kronvang, B. (1992). 'The export of particulate matter, particulate phosphorus and dissolved phosphorus from two agricultural river basins: Implications on estimating the non-point phosphorus load.' *Water Research,* **26**, 1347-1358.

43. Kronvang, B., and Bruhn, A. J. (1996). 'Choice of sampling strategy and estimation method for calculating nitrogen and phosphorus transport in small lowland streams.' *Hydrological Processes* **10**, 1483-1501.

44. Lesack, L. F. W. (1993). 'Export of nutrients and major ionic solutes from a rain forest catchment in the central Amazon Basin.' *Water Resources Research,* **29**, 743-758.

45. Lesack, L. F. W., Hecky, R. E., and Melack, J. M. (1984). 'Transport of carbon, nitrogen, phosphorus, and major solutes in the Gambia River, West Africa.' *Limnology and Oceanography,* **29**, 816-830.

46. Loughran, R. J. (1977). 'Sediment transport from a rural catchment in New South Wales.' *Journal of Hydrology,* **32**, 357-375.

47. Lowrance, R. R., Leonard, R. A., Asmussen, L. E., and Todd, R. L. (1985). 'Nutrient budgets for agricultural watersheds in the southeastern coastal plain.' *Ecology,* **66**, 287-296.

48. Lukatelich, R. J., Schofield, N. J., and McComb, A. J. (1987). 'Nutrient loading and macrophyte growth in Wilson Inlet, a bar-built southwestern Australian estuary.' *Estuarine, Coastal and Shelf Science,* **24**, 141-165.

49. Mattikalli, N. M., and Richards, K. S (1996). 'Estimation of surface water quality changes in response to land use change: Application of the export coefficient model using remote sensing and geographical information system.' *Journal of Environmental Management,* **48**, 263-282.

50. McDowell, W. H., and Asbury, C. E. (1994). 'Export of carbon, nitrogen, and major ions from three tropical mountain watersheds.' *Limnology and Oceanography,* **39**, 111-125.

51. McGarity, J. W., and Munns, D. N. (1955). 'The fertility status of some Kyogle soils and its relation to the soil parent material.' *Journal of the Australian Institute of Agricultural Science,* **September**, 173-178.

52. McKee, L. J., and Eyre, B. D. (1996). 'Nutrient export in a large rurally diverse northern NSW coastal catchment: A preliminary assessment.' In Hunter, H.M., Eyles, A.G., and Rayment, G.E. (eds.). *Downstream Effects of Land Use.* Department of Natural Resources, Queensland, Australia.

53. McMahan, G., and Woodside, M. D. (1997). 'Nutrient mass balance for the Albemarle–Pimlico drainage basin, Northern Carolina and Virginia, 1990.' *Journal of the American Water Resources Association,* **33**, 573-589.

54. Meybeck, M. (1982). 'Carbon, nitrogen, and phosphorus transport by world rivers.' *American Journal of Science,* **282**, 401-450.

55. Meyer, J. L., Likens, G. E., and Sloane, J. (1981). 'Phosphorus, nitrogen, and organic carbon flux in a headwater stream.' *Arch. Hydrobiol*. **91**, 28-44.

56. Milliman, J. D., and Syvitski, J. P. M. (1992). 'Geomorphic/tectonic control of sediment discharge to the ocean: the importance of small mountainous rivers.' *Journal of Geology,* **100**, 525-544.

57. Milliman, J. D. (1995). 'Sediment discharge to the ocean from small mountainous rivers: the New Guinea example.' *Geo-Marine Letter* **15**, 127-133.

58. 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.

59. Moss, A. J., Rayment, G. E., Reilly, N., and Best, E. K. (1993). *A preliminary assessment of sediment and nutrient exports from Queensland coastal catchments.* Environment Technical Report No. 5. Queensland Department of Environment and Heritage and Queensland Department of Primary Industries, Brisbane.

60. Neil, D., and Fogarty, P. (1991). 'Land use and sediment yield on the southern tablelands of New South Wales.' *Australian Journal of Soil and Water Conservation,* **4**, 33-39.

61. 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.

62. Novotny, V., and Chesters, G. (1989). 'Delivery of sediment and pollutants from nonpoint sources: A water quality perspective.' *Journal of Soil and Water Conservation,* **44**, 471-478.

63. Ongley, E. D. (1976). 'Sediment yields and nutrient loadings from Canadian watersheds to Lake Erie: an overview.' *Journal of Fish Research,* **33**, 471-484.

64. Pedrozo, F., and Bonetto, C. (1987). 'Nitrogen and phosphorus transport in the Bermejo River (South America).' *Revue d'Hydrobiologie Tropicale,* **20**, 91-99.

65. Pilleboue, E., and Dorioz, J. M. (1986). 'Mass-balance and transfer mechanisms of phosphorus in a rural watershed of Lac Leman, France.' In Sly, P.G. (ed.). *Sediments and water interactions: proceeding of the third international symposium on interactions between sediments and water.* Geneva, Switzerland, August, 1984. Springer-Verlag, Berlin.

66. Preston, S.D., Bierman Jr, V.J., and Silliman, S.E. (1989) 'An evaluation of methods for the estimation of tributary mass loads.' *Water Resources Research,* **25**, 1379-1389.

67. Pont, D., and Eyre, B. (1997). *Verification of the CMSS nutrient export model in the sub-tropical Brunswick River catchment.* 7th Annual New South Wales Coastal Conference, 'The Coastal Zone—Beyond Beaches', Ballina 12–14 November, 1997.

68. Sharpley, A. N., and Menzel, R. G. (1987). 'The impact of soil and fertiliser phosphorus on the environment.' *Advances in Agronomy,* **41**, 297-324.

69. Singh, K., and Durgunoglu, A. (1989). 'Developing accurate and reliable stream sediment yields.' Sediment and the Environment. *IAHS Publication* **184**, 193-199.

70. Walling, D. E., and Webb, B. W. (1980). 'The spatial dimension in the interpretation of stream solute behaviour.' *Journal of Hydrology,* **47**, 129-149.

71. Walling, D., and Webb, B. (1981). 'The reliability of suspended sediment load data.' Erosion and Sediment Transport Measurement. *IAHS Publication*, **133**, 177-194.

72. Walling, D. E., and Webb, B. W. (1985). 'Estimating the discharge of contaminants to coastal waters by rivers: some cautionary comments.' *Marine Pollution Bulletin,* **16**, 488-492.

73. Walling, D. E., and Webb, B. W. (1988). 'Reliability of rating curve estimates of suspended sediment yield: some further comments.' In Bordas, N.P., and Walling, D.E. (eds*.). 'Sediment budgets.' Proceedings of the Porto Alegre Symposium, IAHS Publication* **174**, 337-350.

74. Walling, D. E., Webb, B. W., and Russell, M. A. (1997). 'Sediment-associated nutrient transport in UK rivers.' In *'Freshwater contamination.' Proceeding of Rabat Symposium S4, IAHS Publication* No. 243. 69-80.

75. Wasson, R. J., Donnelly, T. H., and Murray, A. S. (1996). 'Imports can be dangerous—Appropriate approaches to Australian Rivers and catchments' *First National Conference on Stream Management in Australia* .

76. Wasson, R. J., Mazari, R. K., Starr, B., and Clifton, G. (1998). 'The recent history of erosion and sedimentation on the Southern Tablelands of southeastern Australia: sediment flux dominated by channel incision.' *Geomorphology,* **24**, 291-308.

77. Webb, B. W., and Walling, D. E. (1982). 'Magnitude and frequency characteristics of fluvial transport in a Devon drainage basin and some geomorphological implications.' *Catena,,* **9**, 9-23.

78. Yarbro, L. A., Kuenzler, E. J., Mulholland, P. J., Sniffen, R. P. (1984). 'Effects of stream channelization on exports of nitrogen and phosphorus from North Carolina coastal plain watersheds.' *Environmental Management,* **8**, 151-160.

79. Young, T. C., and DePinto, J. V. (1988). 'Factors affecting the efficiency of some estimators of fluvial total phosphorus load.' *Water Resources Research,* **24**, 1535-1540.

80. 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.

Chapter 3

1. Anon. (1995). *'Investigation of nutrients from urban stormwater and local water quality issues in the Goulburn Broken Catchment.'* **No. 1**, Goulburn Broken Water Quality Working Group.

2. Argent, R. M., and Wilson, C. J. (1996). 'Integrating Research Into Catchment Management.' *23rd Hydrology and Water Resources Symposium*, Hobart, Australia, 471-476.

3. Ball, J., and Trudgill, S. T. (1995). 'Overview of Solute Modelling.' In *Solute Modelling in Catchment Systems.* S. T. Trudgill, ed.. John Wiley and Sons, England, 3-56.

4. Ball, J. E., White, M. J., Innes, G. D., and Chen, L. (1993). 'Application of HSPF on the Upper Nepean Catchment.' *Hydrology and Water Resources Symposium*, Newcastle , NSW, Australia, 343-348.

5. Bennett, J. P. (1974). 'Concepts of mathematical modelling of sediment yield.' *Water Resources Research*, **10**, 485-492.

6. Bernard, G. D. (1993). 'Flood estimation for the Lower Hunter Valley.' *Hydrology and Water Resources Symposium*, Newcastle, Australia, 293-298.

7. Beven, K. (1989). 'Changing ideas in hydrology—the case of physically-based models.' *Journal of Hydrology*, **105**, 157-172.

8. Chapman, B. M. (1991). 'Modelling contaminant transport in natural streams.' *Workshop on Modelling the Fate of Chemicals in the Environment.* I. Moore, ed., Centre for Resource and Environmental Studies, Australian National University, Canberra. 18-21.

9. Cheung, A. S. (1993). 'Modelling the impact of the Oaks Township on water quality of Werriberri Creek.' *International Congress on Modelling and Simulation (MODSIM93)*, Perth , Australia, 275-286.

10. Cheung, A. S., and Fisher, I. H. (1995). 'The use of HSPF program in Total Catchment Management.' *Proceedings of the 16th Federal Convention*, AWWA , Sydney, 2–6 April 1995, Vol. **2**, 747-753.

11. Connolly, R. D., and Silburn, D. M. (1995). 'Distributed parameter hydrology model (ANSWERS) applied to a range of catchment scales using rainfall simulator data. I. Application to Spatially Uniform Catchments.' *Journal of Hydrology*, **172**, 105-125.

12. Connolly, R. D., Silburn, D. M., and Ciesiolka, C. A. A. (1990). 'Distributed parameter hydrology model (ANSWERS) applied to spatially complex catchments using rainfall simulator data.' *Conference on Agricultural Engineering*, Toowoomba, Australia.

13. Connolly, R. D., Silburn, D. M., and Ciesiolka, C. A. A. (1997). 'Distributed parameter hydrology model (ANSWERS) applied to a range of catchment scales using rainfall simulator data. I. Application to a Spatially Complex Catchment.' *Journal of Hydrology*, **193**, 183-203.

14. Davis, J., and Farley, T. (1997). 'CMSS: Policy analysis software for catchment managers.' *Environmental Modelling and Software*, 12.

15. Department of Land and Water Conservation (1995). *'IQQM—Integrated Water Quality and Quantity Model'.* Catchment Processes and Modelling Branch, TS95.019.

16. Dietrich, C., Green, T.R. and Jakeman, A.J (1999). 'An analytical model for stream sediment transport: application to Murray and Murrumbidgee reaches, Australia.' *Hydrological Processes*, **13**(5), 763-776.

17. Dunin, F. (1975). 'Use of physical process models.' *Prediction in Catchment Hydrology—A National Symposium on Hydrology.*, T. Chapman and F. Dunin, eds., Australian Academy of Science, Canberra.

18. Evans, J., and Jakeman, A. (1997). 'Development of a simple, catchment-scale rainfall–evapotranspiration–runoff model.' *Proceedings of the International Congress on Modelling and Simulation, MODSIM97*, **1**, 222-227.

19. Evans, K., Loch, R., Aspinall, T., and Bell, L. (1997). 'Laboratory rainfall simulator studies of selected open-cut mine overburden spoils from Central Queensland.' *Australian Journal of Soil Research*, **35**, 15-29.

20. Evans, K., Loch, R., Silburn, D., Aspinall, T., and Bell, L. (1994). 'Evaluation of the CREAMS model. IV. Derivation of interrill erodibility parameters from laboratory rainfall simulator data and prediction of soil loss under a field rain simulator using the derived parameters.' *Australian Journal of Soil Research*, **32**, 867-878.

21. Evans, K. G., Loch, R. J., Aspinall, T. O., and Bell, L. C. (1992). 'Spoil pile erosion prediction—How far have we advanced?' *Third Large Open Pit Mining Conference*, Mackay, Queensland, 201-205.

22. Fisher, I., Rahman, M., Jivajirajah, J., and Salbe, I. (1993). 'HSPF water quality modelling on South Creek.' *Hydrology and Water Resources Symposium*, Newcastle, New South Wales, Australia, 355-360.

23. Fisher, I. H., and Deen, A. (1993). 'Water quality modelling for a large river basin.' *Proceedings of the International Congress on Modelling and Simulation, MODSIM93*, 263-268.

24. Fisher, P., Abrahart, R., and Herbinger, W. (1997). 'The sensitivity of two distributed non-point-source pollution models to the spatial arrangement of the landscape.' *Hydrological Processes*, **11**, 241-252.

25. Fogarty, P. (1997). 'Testing the Water Erosion Prediction Project (WEPP) model in forest environments in South East NSW.' *Erosion In Forests Workshop.*Coeperative Research Centre for Catchment Hydrology. CSIRO, 4 –6 March 1997..

26. Govers, G., and Loch, R. (1993). 'Effects of initial water content and soil mechanical strength on the runoff erosion resistance of clay soils.' *Australian Journal of Soil Research*, **31**, 549-566.

27. Grayson, R. B., and Doolan, J. M. (1995*).* 'Adaptive Environmental Assessment and Management (AEAM): Notes for Potential AEAM Modellers and Facilitators.' Land and Water RRDC.

28. Grayson, R., Doolan, J., and Blake, T. (1994). 'Application of AEAM (Adaptive Environmental Assessment and Management) to water quality in the Latrobe River catchment.' *Journal of Environmental Management*, **40**, 245-258.

29. Grayson, R., and Moore, I. (1993). 'Effect of Land Surface Configuration on Catchment Hydrology.' *Overland flow: hydraulics and erosion mechanics*, A. Parsons and A. Abrahams, eds., University College London Press Ltd, London, 147-175.

30. Grayson, R. B., Blake, T., and Doolan, J. M. (1993). 'Application of AEAM (Adaptive Environmental Assessment and Management) to water quality in the Latrobe Valley catchment.' *Hydrology and Water Resources Symposium*, Newcastle, Australia.

31. Grayson, R., Moore, I., and McMahon, T. (1992a). 'Physically-based hydrological modelling. I. A Terrain-based model for investigative purposes.' *Water Resources Research*, **28(10)**, 2639-2658.

32. Grayson, R., Moore, I., and McMahon, T. (1992b). 'Physically-based hydrologic modelling: II. Is the concept realistic?' *Water Resources Research*, **28(10)**, 2659-2666.

33. Green, T.R., Beavis, S.G., Dietrich, C.R., and Jakeman, A.J. (1999). 'Relating stream-bank erosion to in-stream transport of suspended sediment' *Hydrological Processes*, **13(5)**, 777-787.

34. Hanley, N., Faichney, R., Munro, A., and Shafe. (1998). 'Economic and environmental modelling for pollution control in an estuary.' *Journal of Environmental Management*, **52**, 211-225.

35. Hatton, T. J., Nicoll, C. L., Hairsine, P. B., and Cresswell, H. P. (1998). 'Models of catchment water quality and their ability to predict the consequences of changes in land-use and management practices.' Farming Action: Catchment Reaction, CSIRO Australia, 321-337.

36. Hook, R. (1997). *Predicting Farm Production and Catchment Processes: A Directory of Australian Modelling Groups and Models.* CSIRO Australia.

37. Jakeman, A.J., Beavis, S.G., Caitcheon, G.G., Green, T.R., Hancock, G., Olley, J.M., Wallbrink, P.J. and Zhang, L. (1999).'A systems approach to determining sediment and phosphorus sources in the Namoi Basin'. *Proceedings of the International Conference on Diffuse Pollution*, Sheraton Perth Hotel, Western Australia, 16–20 May, 1999, 329-341.

38. Jakeman, A.J., Green, T.R., Beavis, S.G., Zhang, L., Dietrich, C.R and Crapper, P.F. (1999). 'Modelling upland and in-stream erosion, sediment and phosphorus transport in a large catchment.' *Hydrological Processes*, **13(5)**, 745-752.

39. Jakeman, A., Post, D., and Beck, M. (1994a). 'From data and theory to environmental model: the case of rainfall runoff.' *Environmetrics*, **5**, 297-314.

40. Jakeman, A., Post, D., Schreider, S. Y., and Ye, W. (1994b). 'Modelling environmental systems: partitioning the water balance at different catchment scales.' Computer Techniques in Environmental Studies V. P. Zannetti, ed., Computational Mechanics Publications, Southampton, 157-170.

41. Jakeman, A.J., and Hornberger, G.M. (1993). 'How much complexity is warranted in a rainfall-runoff model?' *Water Resources Research*, **29(8)**, 2637-2649.

42. Jakeman, A., Littlewood, I., and Whitehead, P. (1990). 'Computation of the instantaneous unit hydrograph and identifiable component flows with application to two small upland catchments.' *Journal of Hydrology*, **117**, 275-300.

43. Kinnell, P. (1998a). 'Applying the USLE-M to erosion grid cells.' *unpublished*.

44. Kinnell, P. (1998b). 'Converting USLE soil erodibilities for use with the $Q_R EI_{30}$ index.' *Soil and Tillage Research*, **45**, 349-357.

45. Kinnell, P. (1996). 'Runoff and sheet erosion from tillage trials under artificial rainfall at Harden, New South Wales.' *Australian Journal of Soil Research*, **34**, 863-77.

46. Kinnell, P. (1993a). 'Interrill erodobilities based on the rainfall intensity flow discharge erosivity factor.' *Australian Journal of Soil Research*, **31**, 319-32.

47. Kinnell, P. (1993b). 'Runoff as a factor influencing experimentally determined interrill erodobilities.' *Australian Journal of Soil Research*, **31**, 333-42.

48. Kinnell, P., and Risse, L. (1998). 'USLE-M: Empirical modelling of rainfall erosion through runoff and sediment concentration.' *Soil. Sci. Soc. Am. J.* (in press).

49. Kleissen, F., Beck, M., and Wheater, H. (1990). 'The identifiability of conceptual hydrochemical models.' *Water Resources Research*, **26(12)**, 2979-2992.

50. Knisel, W. G. (1980). *CREAMS: A Field Scale Model for Chemicals, Runoff and Erosion from Agricultural Management Systems.* USDA, Washington DC.

51. Kwan, R. F., and Abbey, S. (1993). 'Impact of sand and gravel on stability of Leichhardt River at Mt Isa, Queensland.' *Hydrology and Water Resources Symposium*, Newcastle, Australia.

52. Laflen, J. M., Lane, L. J., and Foster, G. R. (1991). 'WEPP: A new generation of erosion prediction technology.' *Journal of Soil and Water Conservation*, **46**, 34-38.

53. Lane, L., Nichols, M., and Paige, G. (1995). 'Modeling erosion on hillslopes: Concepts, theory and data.' *International Congress on Modelling and Simulation Proceedings* (Agriculture, Catchment Hydrology and Industry) **1**, 1-7.

54. Lane, L. J., Renard, K. G., Foster, G. R., and Laflen, J. M. (1992). 'Development and application of modern soil erosion prediction technology—The USDA Experience.' *Australian Journal of Soil Research*, **30**, 893-912.

55. Littleboy, M., Cogle, A., Smith, G., Yule, D., and Rao, K. (1996). 'Soil management and production of Alfisols in the semi-arid tropics I. Modelling the effects of soil management on runoff and soil erosion.' *Australian Journal of Soil Research*, **34**, 91-102.

56. Littleboy, M., Freebairn, D. M., Hammer, G. L., and Silburn, D. M. (1992a). 'Impact of soil erosion on production and erosion risks for a wheat cropping system.' *Australian Journal of Soil Research*, **30**, 775-788.

57. Littleboy, M., Silburn, M. D., Freebairn, D. M., Woodruff, D. R., Hammer, G. L., and Leslie, J. K. (1992b). 'Impact of soil erosion on production in cropping systems. I. Development and validation of a simulation model.' *Australian Journal of Soil Research*, **30**, 757-774.

58. Liu, B., Nearing, M., Baffaut, C., and Ascough, J. (1997). 'The WEPP Watershed Model: III. Comparisons to measured data from a small watershed.' http://danpatch.ecn.purdue.edu/~nearing/wat3.htm.

59. Loague, K., and Freeze, R. (1985). 'A comparison of rainfall-runoff modelling techniques on small upland catchments.' *Water Resources Research*, **21**, 229-248.

60. Loch, R., and Rosewell, C. (1992). 'Laboratory methods for measurement of soil erodibilities (K factors) for the Universal Soil Loss Equation.' *Australian Journal of Soil Research*, **30**, 233-248.

61. Loch, R., Silburn, D., and Freebairn, D. (1989a). 'Evaluation of the CREAMS Model. II. Use of rainulator data to derive soil erodibility parameters and prediction of field soil losses using derived parameters.' *Australian Journal of Soil Research*, **27**, 563-76.

62. Loch, R. J., Maroulis, J. C., and Silburn, D. M. (1989b). 'Rill erosion of a self-mulching Black Earth. II Comparison of sediment transport equations.' *Australian Journal of Soil Research*, **27**, 535-42.

63. Long, G., and Verhoeven, T. J. (1995). 'Nutrient management planning to improve water quality.' *Proceedings 16th Federal Convention of the Australian Water and Wastewater Association, Sydney, 2-6 April 1995.*., 905-908.

64. McQuade, C. V., Collette, G. D., Maschmedt, D. J., and Jolley, D. (1986). 'Application of the ANSWERS runoff/erosion model to catchments in the Adelaide Hills.' *Hydrology and Water Resources Symposium*, Brisbane, Australia, 219-222.

65. Moore, I., and Gallant, J. (1991). 'Overview of hydrologic and water quality modelling.' *Modelling the Fate of Chemicals in the Environment*. I. Moore, ed., Centre for Resource and Environmental Studies, Australian National University, Canberra, 1-8.

66. Moore, I., and Grayson, R. (1991). 'Terrain-based catchment partitioning and runoff prediction using vector elevation data.' *Water Resources Research*, **27(6)**, 1177-1191.

67. Moore, I., Grayson, R., and Ladson, A. (1991). 'Digital terrain modelling: A review of hydrological, geomorphological and biological applications.' *Hydrological Processes*, **5**, 3-30.

68. Nearing, M. A., Lane, L. J., and Lopes, V. L. (1994). 'Modelling soil erosion'. in *Soil Erosion: Research Methods*, R. Lad ed., St. Lucie Press, Florida, 127-156.

69. Phillips, B. C., Blaik S, and Tilley, J. (1993). 'Modelling the effects of land-use change on receiving waters.' *Hydrology and Water Resources Symposium*, Newcastle, Australia, 39-44.

70. Porter, A., and Foster, N. (1996). 'Nutrient management in the Barwon region of New South Wales.' *23rd Hydrology and Water Resources Symposium*, Hobart, Australia, 691-692.

71. Rahman, M., and Salbe, I. (1993). 'Modelling impacts of diffuse and point source nutrients on the water quality of South Creek Catchment.' *International Congress on Modelling and Simulation*, Perth, Australia, 281-287.

72. Renard, K. G., and Ferreira, V. A. (1993). 'RUSLE Model description and database sensitivity.' *Journal of Environmental Quality*, **22**, 458-466.

73. Renard, K. G., Laflen, J. M., Foster, G. R., and McCool, D. K. (1994). 'The Revised Universal Soil Loss Equation.' in *Soil Erosion: Research Methods*, R. Lad, ed., St. Lucie Press, Florida, 105-126.

74. Rizzoli, A. E., and Young, W. J. (1997). 'Delivering Environmental Decision Support Systems—Software tools and techniques.' *Environmental Modelling and Software*, 237-250.

75. Rose, C., and Ghadiri, H. (1991). 'Transport and enrichment of soil-sorbed chemicals.' *Modelling the Fate of Chemicals in the Environment*, I. Moore, ed., Centre for Resource and Environmental Studies, Australian National University, Canberra, 90-101.

76. Rosewell, C. J. (1997). 'Estimation of the RUSLE Cover Management Factor (C) for forest snig tracks.' *Erosion in Forests Workshop.* Cooperative Research Centre for Catchment Hydrology, CSIRO, 4 – 6 March 1997.

77. Rosewell, C. (1995). 'Application of models for evaluating non-point source water pollution in agricultural catchments.' *International Congress on Modelling and Simulation.* (Agriculture, Catchment Hydrology and Industry), **1**, 155-157.

78. Rosewell, C., and Lang, R. (1996). 'Erosion hazard assessment for forest logging operations.' *23rd Hydrology and Water Resources Symposium*, 719-20.

79. Rosewell, C., and Turner, J.B. (1992). 'Rainfall Erosivity in New South Wales.' Technical Report No. 20, Department of Conservation and Land Management, Sydney, NSW.

80. Ryan, P. J., and McKenzie, N. J. (1997). 'Digital terrain attributes and erosion modelling for forests.' *Erosion in Forests Workshop.* Cooperative Research Centre for Catchment Hydrology, CSIRO, 4 – 6 March 1997.

81. Silburn, D. M., and Connolly, R. D. (1995). 'Distributed parameter hydrology model (ANSWERS) applied to a range of catchment scales using rainfall simulator data. I. Infiltration modelling and parameter measurement.' *Journal of Hydrology*, **172**, 87-104.

82. Silburn, D., and Loch, R. (1991). 'Evaluation of the CREAMS erosion model for predicting sediment yields and size distributions.' *Workshop on Modelling the Fate of Chemicals in the Environment.* Centre for Resource and Environmental Studies, Australian National University, Canberra, 141-142.

83. Silburn, D., and Loch, R. (1989). 'Evaluation of the CREAMS model. I. Sensitivity analysis of the soil erosion/sedimentation component for aggregated clay soils.' *Australian Journal of Soil Research*, **27**, 545-61.

84. Silburn, D. M., Connolly, R. D., and Glanville, S. (1990). 'Processes of infiltration under rain on structurally unstable cultivated soils and prediction using a Green and Ampt Model.' *Conference on Agricultural Engineering*, Toowoomba, Australia.

85. Simons, M. Podger, G., Cooke, R. (1996). 'IQQM—A hydrologic modelling tool for water resource and salinity management' *Environmental Software*, **11**, no. 1-3.

86. Spear, R. C. (1995). 'Large simulation models: Calibration, uniqueness and goodness of fit.' *International Congress on Modelling and Simulation Proceedings*, (Agriculture, Catchment Hydrology and Industry), **1,** 8-15.

87. Takken, I., Beuselinck, L., Nachtergaele, J., Govers, G., Poesen, J., and Degraer, G. 'Spatial evaluation of a physically-based distributed erosion model (LISEM).' *Catena, Supplement*. [in press]

88. Titmarsh, G. W., Connolly, R. D., and McLatchey, J. F. (1990). 'Predicting effects of cropping patterns on flood discharges.' *Conference on Agricultural Engineering*, Toowoomba, Australia.

89. Verhoeven, T. J. (1995). 'New South Wales Algal Management Strategy—delivering the vision.' *Proceedings 16th Federal Convention. Australian Water and Wastewater Association*, Sydney, 2–6 April 1995, 205-220.

90. Viney, N., and Sivapalan, M. (1997). 'Modelling sediment loads in the Swan/Avon River Basins.' *Proceedings of the International Congress on Modelling and Simulation, MODSIM97*, **1**, 391-396.

91. Walden, W. J., and Brodie, I. M. (1995). 'Water quality modelling of Trinity Inlet and its catchment.' *Proceedings 16th Federal Convention of the Australian water and Wastewater Association*, Sydney, 2-6 April 1995.

92. Walton, R., and Hunter, H. (1996). 'Modelling water quality and nutrient fluxes in the Johnstone River Catchment, North Queensland.' *23rd Hydrology and Resources Symposium*, Sydney.

93. WBM Oceanics-SKM. (1997). 'Final Report: Task PL2 Phase 1—NonPoint Source (NPS) Catchment Runoff Loads.'

94. Western, A. W., McMahon, T. A., Finlayson, B. L., and O'Neill, I. (1993). 'Wimmera River: Hydrology data and modelling.' *Hydrology and Water Resource Symposium*, Newcastle, Australia, 59-65.

95. Zhang, L., ONeill, A., and Lacy, S. (1995). 'Spatial analysis of soil erosion in catchments: a review of modelling approaches.' *International Congress on Modelling and simulation (MODSIM95)*, Water Resources and Ecology, **3**, 58-64.

Chapter 4

1. Aitken, A. P. (1973). 'Assessing systematic errors in rainfall-runoff models.' *Journal of Hydrology*, **20**, 121-136.

2. Chiew, F., M. Stewardson, and T.A. McMahon (1993). 'Comparison of six rainfall-runoff modelling approaches.' *Journal of Hydrology*, **147**, 1-36.

3. Govers, G., and Loch, R. (1993). 'Effects of initial water content and soil mechanical strength on the runoff erosion resistance of clay soils.' *Australian Journal of Soil Research*, **31**, 549-566.

4. Grayson, R., and Moore, I. (1993). 'Effect of land surface configuration on catchment hydrology.' *Overland Flow: Hydraulics and Erosion Mechanics*, A. Parsons and A. Abrahams, eds., University College London Press Ltd, London, 147-175.

5. Hairsine, P., and Rose, C. (1992). 'Modelling water erosion due to overland flow using physical principles: 1. Sheet flow.' *Water Resources Research*, **28(1)**, 237-293.

6. Hook, R. (1997). *Predicting Farm Production and Catchment Processes: A Directory of Australian Modelling Groups and Models.* CSIRO Australia.

7. Jakeman, A., Green, T., Dietrich, C., Zhang, L., Beavis, S., and Crapper, P. (1997). 'A modelling framework for erosion, sediment and nutrient transport from

catchment to basin scale.' *Proceedings of the International Conference on Modelling and Simulation (MODSIM97)*, **1**, Hobart, Tasmania.

8. Johnston, P. and D. Pilgrim (1976). 'Parameter optimisation for watershed models.' *Water Resources Research.* **12(3)**, 477-486.

9. Lane, L. J., Renard, K.G., Foster, G.R., and J.M. Laflen (1992). 'Development and application of modern soil erosion prediction technology—The USDA experience.' *Australian Journal of Soil Research,* **30**, 893-912.

10. Littleboy, M., Silburn, M. D., Freebairn, D. M., Woodruff, D. R., Hammer, G. L., and Leslie, J. K. (1992). 'Impact of soil erosion on production in cropping systems. I. Development and validation of a simulation model.' *Australian Journal of Soil Research*, **30**, 757-774.

11. Moore, I., Grayson, R., and Ladson, A. (1991). 'Digital terrain modelling: A review of hydrological, geomorphological and biological applications.' *Hydrological Processes*, **5**, 3-30.

12. Moss, A. J., Rayment, G. E., Reilly, N., and Best, E. K. (1993). A Preliminary Assessment of Sediment and Nutrient Exports from Queensland Coastal Catchments. *Environment Technical Report No. 5.* Queensland Department of Environment and Heritage and Queensland Department of Primary Industries, Brisbane.

13. Rizzoli, A. E., and Young, W. J. (1997). 'Delivering environmental decision support systems—Software tools and techniques.' *Environmental Modelling and Software*, 237-250.

14. Sorooshian, S. and V. K. Gupta (1983). 'Automatic calibration of conceptual rainfall-runoff models: the question of parameter observability and uniqueness.' *Water Resources Research*, **19(1)**, 1260-1268.

APPENDIX I

1. Baginska, B., Cornish, P.S., Hollinger, E., Kuczera, G. and Jones, D. (1998). 'Nutrient export from rural land in the Hawkesbury–Nepean Catchment' *Proceedings of the 9th Australian Agronomy Conference, Wagga Wagga*, 753-756.

2. Bek, P., and Bruton, G. (1979). 'Gippsland Lakes and Streams Hydrochemistry.' Ministry of Conservation Studies Services, Melbourne.

3. Birch, P. (1982). 'Phosphorus export from Coastal Plain discharge into the Peel–Harvey Estuarine System of Western Australia.' *Australian Journal of Freshwater Research*, **33**, 23-32.

4. Bott, G. (1993). '*Relationships Between the Extent of Conventional, Broadscale Agriculture and Stream Phosphorus Concentration and Phosphorus Export in South-West Australia.'* Office of Catchment Management, Perth, Western Australia.

5. Chittleborough, D. (1983). 'The nutrient load in surface waters as influenced by land use patterns.' *The Effects of Change in Land Use Upon Water Resources*, J. Holmes, ed., Water Research Foundation of Australia, Adelaide.

6. Clarke, R. (1988). *The Export of Nitrogen, Phosphorus and Turbidity from Catchments with Various Land Uses in the Mt Lofty Ranges of South Australia. EWS Library Reference 87/17*, Engineering and Water Supply Department, South Australia.

7. Commission, N. C. D. (1978). *ACT Region Water Quality Study*. Australian Department of Construction, Binnie International (Australia) Pty. Ltd.

8. Cosser, P. (1989). 'Nutrient concentration – flow relationships and loads in the South Pine River, Southern Tablelands, New South Wales.' *Australian Journal of Marine and Freshwater Research*, **40**, 613-630.

9. Costin, A. (1980). 'Runoff and soil nutrient losses from an improved pasture at Ginninderra, Southern Tablelands, New South Wales.' *Australian Journal of Agricultural Research*, **31**, 533-546.

10. Cullen, P. (1991). *Regional Catchment Management and Receiving Water Quality: The Monkey Creek Project Final Report.* Report to the Land and Water Resources RDC, Sydney.

11. Cullen, P. (1995). 'Managing non-point sources of phosphorus from rural areas.' *Water*, May/June, **13**.

12. Cullen, P., Farmer, N., and O'Loughlin, E. (1988). 'Estimating non-point sources of phosphorus to lakes.' *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie*, **23**, 588-593.

13. Cullen, P., and Rosich, R. (1979). 'Effects of rural and urban sources of phosphorus of Lake Burley Griffin.' *Progress in Water Technology*, **11**, 219-230.

14. Cullen, P., Rosich, R., and Bek, P. (1978). *A Phosphorus Budget for Lake Burley Griffin and Management Implications for Urban Lakes. Tech. Paper No. 31,* AWRC Canberra.

15. Department of Natural Resources (1997). *From Land to River to Reef Lagoon: Land Use Impacts on Water Quality in the Johnstone River Catchment.* DNR, Brisbane.

16. Dillon, P., and Molot, L. (1997). 'Effect of landscape form on export of dissolved organic carbon, iron and phosphorus from forested stream catchments.' *Water Resources Research*, **33(11)**, 2591-2600.

17. Envirotest. (1996). *Water Quality—Brisbane River Catchment and Moreton Bay, Stage Two Foundation Paper.* Brisbane River Management Group.

18. Forbes, G., and Birch, P. (1987). 'Nutrient loading in the Peel–Harvey Estuary from the Harvey, Murray and Serpentine Rivers (1977–1984).' *Bulletin 182*, Department of Conservation and Environment, Western Australia.

19. Gourley, J., Cogle, A.L., Brebber, L., Herbert, B., Best, E. and Wright, N. (1996) 'Water quality and land use impacts on Lake Tinaroo/Barron River: IV. Decision Support Systems for use in the Tinaroo Dam Catchment.' In H.M. Hunter, A.G. Eyles, G.E. Rayment (eds.) *Downstream Effects of Land Use.* Department of Natural Resources, Queensland, Australia. 261-264.

20. Gutteridge, Haskins and Davey and EPA Victoria (1981). 'Characterisation of pollution in urban stormwater.' *Technical Paper No. 60*, AWRC.

21. Lepisto, A. (1995). 'Increased leaching of nitrate at two forested catchments in Finland over a period of 25 years.' *Journal of Hydrology*, **171**, 103-123.

22. Marston, F., Young, W., and Davis, J. (1995). *Nutrient Generation Databook, Second Edition*, Unpublished CMSS Reference Manual, CSIRO Division of Water Resources, Canberra, Australia.

23. SKM and WBM Oceanics (1998). *Brisbane River and Moreton Bay Wastewater Management Study*. Task PL2—Phase 2.

24. Moss, A., Rayment, G., Reilly, N., and Best, E. (1992). Sediment and Nutrient Exports from Queensland Coastal Catchments. *Unpublished Report*, Department of Environment and Heritage and Department of Primary Industries, Brisbane.

25. Prove, B., and Hicks, W. (1991). 'Soil and nutrient movements from rural land of Northern Queensland.' *Land Use Patterns and Nutrient Loading of the Great Barrier Reef Region: Proceedings of a Workshop*, James Cook University, North Queensland.

26. Smalls, I. (1986). 'Stormwater runoff and some environmental implications in the Sydney Region.' *Proceedings of the 12th Symposium, Storm water Quality in Urban Areas*, Illawarra Regional Committee, Wollongong.

27. Tan, H. (1991). *Nutrients in Perth Urban Surface Drainage Catchments Characterised by Applicable Attributes*. *Report No. WS 85*, WAWA, Perth.

28. Wood, G. (1986). *Mt Lofty Ranges Watershed: Impact of Land Use on Water Quality and Implications for Reservoir Water Quality Management.* EWS Library Reference 86/19, Engineering and Water Supply, South Australia.

29. Young, W., Marston, F., and Davis, J. (1997). 'NEXSYS: An expert system for estimating non-point source nutrient export rates.' *AI Applications*, **11(2)**, 11-18.

30. Young, W., Marston, F., and Davis, J. (1996). 'Nutrient exports and land use in Australian catchments.' *Journal of Environmental Management*, **47**, 165-183.

APPENDIX II

Hook, R. (1997). *Predicting Farm Production and Catchment Processes: A Directory of Australian Modelling Groups and Models.* CSIRO Australia.

APPENDIX I NUTRIENT GENERATION RATES

APPENDIX I Nutrient Generation Rates

Literature on nutrient and sediment export rates in Australia is fairly sparse. Many Australian studies using nutrient generation and export rates rely upon rates quoted from North American data. The use of such data in Australian conditions is likely to lead to errors in the assessed sediment and nutrient loads in catchments for various land uses, as North American export rates are often higher than corresponding Australian export rates, due to differences between Australian and North American land use and land management practices, climate, soils and vegetative structure. Young *et al.* (1996) completed a comprehensive review of Australian and North American literature on nutrient export rates, emphasising these difficulties in transferring results from North American studies to Australian conditions.

Table A.1: Nutrient export rates as reported in Young *et al.* (1996)

Broad land use type	Total Phosphorus (kg/ha/yr)		Total Nitrogen (kg/ha/yr)	
	Range	Typical	Range	Typical
Urban				
South-eastern Australia	0.4–3.6	1.0	3.2–22.4	6.6
North-eastern Australia	–	–	–	–
Western Australia	0.1–1.1	0.4	1.0–6.6	2.5
Improved Pasture				
South-eastern Australia	0.1–0.7	0.3	0.6–4.6	3.3
North-eastern Australia	0.25–1.0	0.5	3.4–10.8	7.5
Western Australia	0.5–0.9	1.1	2.4–3.5	3.0
Unimproved Pasture				
South-eastern Australia	0.07	0.07	2.2	2.2
North-eastern Australia	0.05–0.08	0.06	2.7–5.1	3.5
Western Australia	0.002–0.4	0.1	–	–
Cropping				
South-eastern Australia	–	–	–	–
North-eastern Australia	1.9	1.9	12.3	12.3
Western Australia	–	–	–	–
Market Gardens				
South-eastern Australia	2.7–14.3	7.1	20–34.5	26
North-eastern Australia	–	–	–	–
Western Australia	–	–	–	–
Forests				
South-eastern Australia	0.03–0.1	0.06	0.9–1.5	1.1
North-eastern Australia	0.1–0.2	0.14	0.9	0.9
Western Australia	0.001–0.1	0.05	–	–

As shown in Table A.1, Young *et al.* (1996) compiled tables of nutrient export rates for six broad land use types in south-eastern, western and north-eastern Australia. These reported export rates generally consist of ranges of values and typical values for total nitrogen and total phosphorus export. Nutrient export rates for south-eastern Australia were taken from Bek and Bruton (1979), Chittleborough (1983), Cullen (1991), Cullen *et al.* (1978, 1988), Cullen and Rosich (1979), Clark (1988), Costin (1980), Smalls (1986), Wood (1986), GHD (1981) and NCDC (1978). Rates for Western Australian land uses were taken from Birch (1982), Bott (1993) Forbes and Birch (1987) and Tan (1991). Rates for north-eastern Australia from Cosser (1989), Moss *et al.* (1992) and Prove and Hicks (1991).

This review effectively covers the majority of Australian literature on nutrient export rates up to 1996, and has been used in the construction of the NEXSYS program, a simple rule-based expert system for the estimation of non-point source nutrient export rates for the CMSS program (Young *et al.* 1997).

I.1 NEXSYS

NEXSYS was developed by CSIRO Land and Water for use with the CMSS model. It can be used to provide ranges of nutrient export rates for various land use and land management options. These ranges are estimated from export rates reported in Australian studies on nutrient export and are provided in a form compatible with use in the CMSS model.

NEXSYS is a simple rule-based expert system. The user is queried for information on various land management and environmental factors. NEXSYS works by classifying land use into five broad land use types: urban, grazing, cropping, forests and horticulture. When the user specifies only land use and nutrient type, NEXSYS provides a full range of export rates. As further information describing environmental and management factors is input, narrower ranges of export rates are given. NEXSYS never suggests a single value for nutrient export rates; rather, it always reports a range of values. This reflects the natural variability of export rates and the poor level of current understanding in this area. The number of subranges in each land use category depends upon the number of values reported in Australian literature for that land use type. NEXSYS is provided to all users of CMSS software.

I.2 The CMSS Nutrient Generation Handbook

Additional information on nutrient export rates suitable for use in the CMSS model is provided in Marston *et al.* (1995). This is a Data Book containing the results of Australian and overseas studies on nutrient generation rates, categorised both by author and by broad land use category. This Data Book is also provided with the CMSS model software.

I.3 Recent Literature on Nutrient Export Rates

Very few studies on nutrient generation rates have been completed in Australia since 1995–96. Some work has been completed in the Johnstone River catchment near Innisfail in northern Queensland (Department of Natural Resources 1997). In this study the HSPF model was calibrated on data collected between 1991 and 1996. Land use in the catchment was divided into six broad groups: bananas, sugar cane, rainforest, dairy pasture, beef pasture and unsewered residential (including rural residential and unsewered towns). The effects of roads on nutrient export rates was included in the appropriate land use in each case. The model was then simulated on 40 years of observed rainfall data, and the sediment, phosphorus and nitrogen export rates for each land use category calculated per hectare per year. These results are summarised in Table A.2.

Table A.2: Nutrient and sediment export rates in Department of Natural Resources (1997)*

Land use type	Suspended sediment (t/ha/yr)	Total Phosphorus (kg/ha/yr)	Total Nitrogen (kg/ha/yr)
Bananas	4	42	7
Sugar cane	4	39	7
Rainforest	1	10	2
Pasture (dairy)	1	9	2
Pasture (beef)	1	15	2
Unsewered	1	70	2

* all values are approximate

Other nutrient export rates quoted in a recent review of Australian literature (SKM and WBM Oceanics 1998) include values for total phosphorus export for grazing and urban areas (Cullen 1995), and values for total phosphorus and total nitrogen export for rural residential, rural undisturbed, agriculture, urban residential and industrial (Envirotest 1996). These export rates (in kg.ha^{-1}yr^{-1}) are given in Table A.3.

Table A.3: Nutrient export rates in SKM and WBM Oceanics (1998)

Source	Land use type	Total Nitrogen (kg/ha/yr)	Total Phosphorus (kg/ha/yr)
Cullen (1995)	Grazing	–	0.2–0.6
	Urban	–	1.2
Envirotest (1996)	Rural residential	1	0.2
	Rural undisturbed	1.8	0.2
	Rural other	1	0.2
	Agriculture	10	1.5
	Urban residential	10	1.5
	Urban industrial	7.5	1.0

Baginska *et al.* (1998) derived phosphorus export rates for subcatchments of the Hawkesbury–Nepean catchment in western Sydney. Estimates were made using data collected at eleven nested monitoring sites, situated to allow the calculation of contributing nutrient loads from different land uses. The phosphorus loads estimated for different land uses are summarised in Table A.4.

Table A.4: Phosphorus export rates in Baginska *et al.* (1998)

Land use	Phosphorus (kg/ha/yr)
Market Garden	15.3
Intensive Dairy	6.4
Extensive Dairy	1.9–2.5
Semi-improved pasture/hobby	0.8
Unimproved	0.33

Gourley *et al.* (1996) completed a comparison of CMSS and AGNPS on the Tinaroo Dam catchment in northern Queensland. They report a number of nutrient generation rates for different land uses, estimated from a literature review and local knowledge. These nutrient generation rates are shown in Table A.5.

Other recent literature on nutrient export rates includes studies by Lepisto (1995) and Dillon and Molot (1997). Lepisto (1995) quotes nitrogen export rates for two forested catchments in Finland, whilst Dillon and Molot (1997) looked at phosphorus and other nutrient export rates in seven undisturbed, forested catchments in Ontario. Neither of these studies are considered to be applicable to Australian conditions as the climatic and vegetative conditions of the catchments considered in these studies are very different from typical Australian catchments.

Table A.5: Nutrient export rates in Gourley *et al.* (1996)

Land use type	Total Phosphorus (kg/ha/yr)	Total Nitrogen (kg/ha/yr)
Avocado	2	3
Avocado/Macadamia	2	3
Bare Land	3	1.5
Clear Pasture	0.2	1.5
Closed Forest	0.1	1
Cropping	2	4
Dairy	0.2	3
Grazing	0.15	1.5
Macadamia	2	3
Maize	0.3	4
Open Forest	0.1	1.3
Orchard	2	3
Pasture/Scrub	0.15	1.5
Pine Plantation	0.07	1
Poultry farm	5	15
Rainforest	0.13	0.5
Rural Residential	0.7	2
Sewerage Treatment Plant	2318	8584
Swamp	0.1	0.5
Tourist	0.1	0.5
Unsurveyed	0.1	1.8
Urban	1.3	2

I.4 Nutrient Generation Rate Literature Summaries

This appendix includes individual summaries of papers and reports quoting nutrient generation rates that are referenced in section I.3.

Baginska, B., Cornish, P.S., Hollinger, E., Kuczera, G., and Jones, D. (1998) 'Nutrient export from rural land in the Hawkesbury–Nepean catchment', Proceedings of the 9th Australian Agronomy Conference, Wagga Wagga, pp 753–756.

SUMMARY	Nutrient generation rates were estimated for different land uses using a series of nested monitoring stations at three different areas within the catchment. Data collection occurred at eleven such stations for periods of 18–30 months. Computer modelling was used to estimate the annual generation rate over a much longer period (1881–1993).
CATCHMENT CHARACTERISTICS	The Currency creek catchment covers approximately 225 ha in western Sydney. Land use in the area of the study was divided into five broad categories: market garden, intensive dairy, extensive dairy, semi-improved pasture/hobby and unimproved Other monitoring areas were Mangrove Mountain and Camden. Data from Mangrove Mountain was found to be insufficient for the study as significant runoff occurred only twice during the monitoring period. Data for the Camden area were included but were also affected but the small number of runoff events.

GENERATION RATES AND LAND USES

Land Use	**Generation Rates Phosphorus (kg/ha/yr)**
Market Garden	15.3
Intensive Dairy	6.4
Extensive Dairy	1.9–2.5
Semi-improved pasture/hobby	0.8
Unimproved	0.33

Department of Natural Resources (1997) 'From Land to River to Reef Lagoon: Land Use Impacts on Water Quality in the Johnstone River Catchment', DNR, Brisbane.

SUMMARY

The HSPF model was calibrated on data collected between 1991 and 1996. The model was then simulated on 40 years of observed rainfall data, and the suspended sediment, phosphorus and nitrogen export rates for six broad land use categories were calculated per hectare per year.

CATCHMENT CHARACTERISTICS

The Johnstone River catchment covers 1634 km^2, and is situated near Innisfail in Northern Queensland.

Approximately half the catchment is covered by relatively pristine rainforest, while beef and dairy grazing, sugar cane and banana growing are the other dominant land uses.

Most rainfall occurs over the summer months, with mean annual rainfall of 1673 mm at Malanda in the upper catchment and 3545 mm at Innisfail on the coast.

The effect of roads on nutrient generation rates in the catchment were included in the appropriate land use, rather than being accounted for separately.

GENERATION RATES AND LAND USES

Land Use	**Generation Rates**		
	Sediment (T/ha/yr)	**Phosphorus (kg/ha/yr)**	**Nitrogen (kg/ha/yr)**
Bananas	4	42	7
Sugar Cane	4	39	7
Rainforest	1	10	2
Pasture (dairy)	1	9	2
Pasture (beef)	1	15	2
Unsewered	1	70	2

Envirotest (1996) 'Water Quality—Brisbane River Catchment and Moreton Bay Stage Two Foundation Paper', Brisbane River Management Group.

SUMMARY

Envirotest used pollutant generation rates to estimate the total pollutant loads in the Brisbane River catchment, estimating the contributing load of each different land use.

CATCHMENT CHARACTERISTICS

The Brisbane River catchment covers 13560 km^2 and includes a number of subcatchments, namely the Upper Brisbane River, the Stanley river, Lockyer Creek and Bremer River. The catchment covers both urban and rural areas, including the city of Brisbane.

The method used to estimate these nutrient export rates was not detailed in the report.

GENERATION RATES AND LAND USES

Land Use	**Generation Rates**	
	Phosphorus (kg/ha/yr)	**Nitrogen (kg/ha/yr)**
Rural Residential	0.2	1
Rural Undisturbed	0.2	1.8
Rural Other	0.2	1
Agriculture	1.5	10
Urban Residential	1.5	10
Urban Industrial	1.0	7.5

Gourley, J., A.L Cogle, L. Brebber, B. Herbert, E. Best and N. Wright (1996) 'Water Quality and Land Uses on Lake Tinaroo/Barron River. IV. Decision Support Systems for use in the Tinaroo Dam Catchment'. In: Downstream Effects of Land Use, pp. 261–264, H.M. Hunter, A.G. Eyles and G.E. Rayment (eds), Department of Natural Resources, Queensland.

SUMMARY

Nutrient generation rates for the Tinaroo catchment for use in the CMSS model are given, based on local knowledge and a literature review.

CATCHMENT CHARACTERISTICS

The Tinaroo Dam is on the Barron River near Cairns in northern Queensland. The catchment covers 54 000 ha with a diverse range of land uses.

Details are not given of the literature surveyed or the sources of local knowledge or extent of its use in this paper.

GENERATION RATES AND LAND USES

Land Use	**Generation Rates**	
	Phosphorus (kg/ha/yr)	**Nitrogen (kg/ha/yr)**
Avocado	2	3
Avocado/Macadamia	2	3
Bare land	3	1.5
Clear Pasture	0.2	1.5
Closed Forest	0.1	1
Cropping	2	4
Dairy	0.2	3
Grazing	0.15	1.5
Macadamia	2	3
Maize	0.3	4
Open Forest	0.1	1.3
Orchard	2	3
Pasture/Scrub	0.15	1.5
Pine Plantation	0.07	1
Poultry Farm	5	15
Rainforest	0.13	0.5
Rural Residential	0.7	2
Sewerage Treatment Plant	2318	8584
Swamp	0.1	0.5
Tourist	0.1	0.5
Unsurveyed	0.1	1.8
Urban	1.3	2

APPENDIX II MODELLING GROUPS IN AUSTRALIA

APPENDIX II Modelling Groups in Australia

A number of groups concerned with modelling water quality and sediment and nutrient transport exist in Australia. This section provides some information on these groups, including contact details for further information. Details on modelling groups within Australia are published in a report compiled by Hook (1997).

II.1 Centre for Catchment and In-Stream Research

The Centre for Catchment and In-Stream Research is based at Griffith University. It is involved in the development of physics-based models for the prediction of soil erosion from hillslopes, including GUEST, and in the development of other computer programs that allow the determination of relevant parameters for use in other models. These models and programs are targeted at land management agencies and researchers in Australia and throughout South-East Asia.

Contact Details:

web site: http://www.ens.gu.edu.au/ecology/ccisr/intro.htm
contact: Emeritus Professor Calvin Rose
Phone: (07) 3875 7397
Email: c.rose@ens.gu.edu.au

II.2 Centre for Integrated Resource Management (CIRM)

CIRM is a cooperative research and education group involving the Queensland Department of Natural Resources, the Queensland Department of Primary Industries and the University of Queensland. The centre is developing models for integrated catchment assessment, including environmental, economic and social issues, and for simulation of the effects of land use/land management practices and climate on the movement of nutrients at various catchment scales. These models are targeted at catchment groups, industry and government agencies.

Contact Details:

web site: http://www.geosp.uq.edu.au/irm/research.htm
contact: Dr David Gramshaw
University of Queensland
Phone: (07) 3365 6879
Email: dgram@cirm.uq.edu.au

II.3 Centre for Resource and Environmental Studies (CRES)

CRES is a research centre within the Australian National University. Areas of concern at CRES include surface water and ground water modelling, in terms of both discharge and water quality, using a wide range of models. Dr Peter Kinnell is working on the development of the USLE-M model, and has experience with sediment and nutrient transport modelling.

Contact Details:

web site: http://cres.anu.edu.au
contact: Professor Tony Jakeman
Phone: (02) 6249 4742
Email: tony@cres.anu.edu.au

II.4 Centre for Water Research (CWR)

The Centre for Water Research is located at the University of Western Australia, Perth, and is involved in a number of projects involving water quality measurements and modelling. Models being used and investigated at the Centre include LASCAM, a model for the simulation of water yield and salinity in large catchments; an urban surface catchment model; ELCOM-2D, a water quality model for lakes and reservoirs; and QUAL2D, a river hydrodynamics and water quality model.

Catchment work being undertaken at CWR includes water balance modelling, modelling of erosion and sediment transport, and modelling of the effects of changes in land use on water quality.

Contact Details:

web site: http://www.cwr.uwa.edu.au
contact: Associate Professor M. Sivapalan
University of Western Australia
Phone: (08) 9380 2320
E-mail: sivapalan@cwr.uwa.edu.au

II.5 CSIRO Land and Water

CSIRO Land and Water specialises in research on soil, water and atmospheric processes essential to the understanding and sustainable management of land and water resources in Australia and internationally. CSIRO Land and Water has staff based in Perth, Adelaide, Canberra, Albury, Griffith, Brisbane, Townsville and Atherton. The Division is involved in a number of areas of research including sediment, nutrient and pollutant transport in catchments and surface water management. Models used and developed within the Division include CMSS and TOPOG.

Contact Details:

web site: http://www.clw.csiro.au/division/
contact: Dr Richard Davis
Phone: (02) 6246 5706
Email: richard.davis@cbr.clw.csiro.au

Dr Rob Vertessy
Phone: (02) 6246 5790
Email: rav@cbr.dwr.csiro.au

II.6 CRC for Catchment Hydrology

The Cooperative Research Centre for Catchment Hydrology is a partnership between the University of Melbourne, Monash University, CSIRO Division of Water Resources, Bureau of Meteorology, Department of Conservation and Natural Resources, Melbourne Water, Murray–Darling Basin Commission, and the Rural Water Corporation of Victoria. Other participating organisations include Brisbane City Council, Sydney Water and the Water Services Association of Australia.

The Centre carries out research in four program areas:

- water, vegetation and solutes
- soil erosion and channel stability
- urban hydrology and environmental flow management
- floods and hydrology regionalisation.

The Centre also has a program dealing with education and training.

Contact Details:

web site: http://www.catchment.crc.org.au/index.htm
contact: Dr. Francis Chiew
University of Melbourne
Phone: (03) 9344 6644
Email: fchs@engineering.unimelb.edu.au

Prof. Tom McMahon
University of Melbourne
Phone: (03) 9344 6641
Email: tam@engineering.unimelb.edu.au

II.7 Department of Land and Water Conservation, NSW

The Department of Land and Water Conservation, NSW, through its integrated water quantity and quality model (IQQM) project, is involved in the development of a conceptual model, IQQM, designed to address water quality and environmental issues. The model includes components for modelling the nitrogen and phosphorus cycles, dissolved oxygen and algae, and other water quality factors. The model is intended for the evaluation of policies for water sharing and other water resource management options, predominantly in the Murray–Darling basin. It is targeted at water resource managers, hydrologists, water users and environmentalists.

Contact Details:

web site: http://www.dlwc.nsw.gov.au/
contact: Mr Geoff Podger
Phone: (02) 9895 7480
Email: gpodger@dlwc.nsw.gov.au

II.8 Griffith University and CSIRO Land and Water

Griffith University with CSIRO (Peter Hairsine) are involved in a project to develop algorithms to model the flush of fine sediment during erosion. These algorithms consist of a set of physically based equations describing erosion processes on a hillslope. These are targeted at researchers in soil erosion and water quality.

Contact Details:

contact: Professor Bill Hogarth
Griffith University
Phone: (07) 3875 7430

II.9 Hydrotech Research Pty Ltd

Hydrotech Research is a software distributor for the Danish Hydraulic Institute, and is involved in model development related to DHI models and to codes developed by the company. The company is also involved in data preparation, model calibration and simulations. The models used by Hydrotech Research include hydrodynamic models such as MIKE-SHE and MIKE-11, as well as conceptual rainfall runoff models such as NAM and RORB. These models are targeted at research organisations, public organisations and consultants.

Contact Details:

contact: Dr Robert Carr
Phone: (02) 9955 4030

II.10 Integrated Catchment Assessment and Management Centre (ICAM)

ICAM is a centre established within the Australian National University's School of Resource Management and Environmental Science. It has the ability to draw from a wide range of researchers within the ANU, such as the Centre for Resource and Environmental Studies, and from other external organisations. The focus of ICAM is a multidisciplinary approach to catchment management, including modelling of biophysical factors in catchment management such as surface water modelling, in terms of quality and quantity of water discharged, and erosion.

Contact Details:

web site: http://cres.anu.edu.au/icam/
contact: Mr Chris Buller
Phone: (02) 6249 3568
Email: buller@cres.anu.edu.au

II.11 International Association on Water Quality (IAWQ)

IAWQ currently has 35 specialist groups, each of which each of which acts like a technical division with its own leadership. The Diffuse (Non-Point) Source Pollution Group is concerned with a wide range of issues to do with pollution and water quality, including modelling of sediment and nutrient transport and water quality in catchments.

Contact Details:

web site: http://www.iawq.org.uk/spgroups/index.htm
contact: Mr Lance Bowen
Centre for Wastewater Treatment
University of NSW
Phone: (02) 9385 5047
E-mail: cwwt@civeng.unsw.edu.au

II.12 Modelling and Simulation Society of Australia and New Zealand Inc.

The Modelling and Simulation Society of Australia and New Zealand Inc. (MSSANZ), formerly the Modelling and Simulation Society of Australia (MSSA) and the Simulation Society of Australia Inc. (SSA), is an affiliate of the International Association for Mathematics and Computers in Simulation (IMACS). The aims of the Society are to promote, develop and assist in the study of all areas of modelling and simulation. Members of the Society include professional hydrologists and others interested in catchment modelling.

Contact Details:

web site: http://cres.anu.edu.au/~tony/mssanz
contact: Professor Anthony Jakeman
Centre for Resource and Environmental Studies
Australian National University
Phone: (02) 6249 4742
Email: tony@cres.anu.edu.au

II.13 Sinclair Knight Merz (SKM)

Sinclair Knight Merz is a multi-disciplinary firm of consultant engineers, planners and scientists. SKM develop models to estimate catchment yield and simulate catchment processes and runoff quality. The types of models developed by SKM include conceptual rainfall runoff models, physics-based models, models for use in AEAM and distributed conceptual models of catchment water balance and salinity processes. These models have been developed for use by engineering hydrologists and modellers to provide advice to a range of community and government groups.

Contact Details:

web site: http://www.skm.com.au
contact: Dr Rory Nathan
Phone: (03) 9248 3322
Email: rnathan@skm.com.au

II.14 Unisearch Water Research Laboratory (UWRL)

Unisearch Water Research Laboratory is based in the University of New South Wales. UWRL is capable of 1-D, 2-D and 3-D hydrodynamic and water quality modelling of coastal and estuarine areas, catchment runoff, pollutant source identification, and sediment transport. It has access to a range of software, including commercial, public-domain, academic and in-house models.

Contact Details:

web site: http://www.unsw.edu.au/unisearch/wrl
contact: Phone: (02) 9949 4488
Email: office@manly.civeng.unsw.edu.au

II.15 University of Melbourne

This group of modellers consists of the Centre for Environmental Applied Hydrology an the CRC for Catchment Hydrology. The University of Melbourne is involved in modelling rainfall runoff and water quality; in providing tools for catchment management; and in the development of a wide range of empirical, conceptual and physics-based models, including THALES, MODHYDROLOG and models for use in AEAM, targeted at researchers, engineers, land managers and community groups.

Contact Details:

web site: http://www.civeng.unimelb.edu.au/ceah/ceah.html
contact: Dr Rodger Grayson
Phone: (03) 9344 7305
Email: rodger@civag.unimelb.edu.au

II.16 WBM Oceanics

WBM Oceanics is a Brisbane-based environmental and engineering consultancy firm. WBM Oceanics is involved in projects to simulate and predict the effects on river water quality of proposed treatment and management scenarios, implementing the MIKE11 model.

II.17 Waters and Rivers Commission, WA

The Water and Rivers Commission is developing models to simulate physical processes within the water cycle using empirical relationships. These models, such as MAGIC, are spatially distributed and are targeted for use by planning groups in natural resource management agencies.

Contact Details:

web site: http://www.wrc.wa.gov.au
contacts: Dr Malcolm Robb
Phone: (08) 9278 0526
Email: malcolm.robb@wrc.wa.gov.au

Mr G.W. Mauger
Phone: (08) 9278 0472
Email: geoff.mauger@wrc.wa.gov.au