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Catchment Phosphorus Sources and Algal Blooms — An Interpretative Review

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ABSTRACT

Australia's water quality problems with toxic blue-green algal blooms were brought to a focus with the occurrence of these blooms, in 1991, extending over 1000 km of the Darling-Barwon River. Public and political pressure demanded quick remedial actions in an environment where there was little understanding of how to carry out remedial work at the catchment scale. The problem with hasty remedial action is that it may lead to disappointing results over time, distrust and frustration with scientific research, and with the methods of restoration. It is easy to focus on point sources because they are easily identified, measured, and susceptible to control by policies and regulation. However, unless we have some understanding of the important parts of the processes leading to algal bloom formation in Australia, and at least qualitative mass balances of loading to waterways, the very large amounts of money being used to try and decrease the frequency of algal blooms may be wrongly targeted.

In this study of catchment phosphorus (P) sources and algal blooms we review the existing literature and work in progress, as it applies to our understanding of the movement of P from landscapes into and along stream systems of basins typified by the Murray-Darling Basin (i.e. agriculture/grazing on clay-rich soils - a large part of Australia). The uniqueness of the Australian environment, which caused Harris (1995) to comment that "uncritical importing of overseas research and management techniques is not necessarily the best option", is highlighted and put into context with our understanding of the bioavailability of P in Australian environments. We examine the processes/mechanisms of P movement through Australian landscapes and the possibility of developing a P budget. The penultimate section of this document is a quest for a general catchment model of P sources which draws on discussion of the two previous sections, and describes the limitations in the present models that claim to predict the effect of land use change on P exports to waterways.

INTRODUCTION

The occurrence in late 1991 of what is claimed to be the world's largest riverine blue-green algal bloom, along the Darling-Barwon River caused both Federal and State Governments to react to a concern that toxic cyanobacterial blooms were increasing in frequency in rural Australia. Water quality managers have accepted that the limiting nutrient necessary for the growth of cyanobacterial (blue-green algal) blooms in inland water systems is phosphorus (P) and therefore management of the problem has been based on the premise that phosphorus-load reductions to waterways is the key control factor (Algal Management Strategy, 1993). When P levels are high in Australia's turbid water systems, blue-green algae can gain competitive advantages because of: (1) their ability to overcome the spatial separation between light and nutrients through the use of gas vacuoles (intracellular gas-filled spaces), and (2) the ability of many of these species to fix molecular nitrogen and therefore dominate in waterways with low N:P ratios (e.g. where eutrophication and denitrification has caused loss of N_2).

Discussions of the sources of P in the Australian aquatic environment have, however, tended to accept as axiomatic that the predominant sources are some combination of sewage, fertiliser, feed-lot effluent, detergents, etc. This assumption has the convenience that, at least in principle, what has been done can be undone - that in some way it should be possible to return Australian waterways to some earlier 'pristine' (oligotrophic) condition. This axiom undoubtedly arose because of its broad truth in the more heavily populated and/or industrialised parts of the world, particularly in Europe and North America. There the problems of eutrophication have been actively studied for decades, and there is considerable evidence for increases in phosphate concentrations as a result of human activities (e.g. Morse et al., 1993).

There are certainly site-specific examples of human impact in Australia. For example, in Port Phillip Bay, Victoria (Hall, 1992), and in the Peel-Harvey estuarine system of Western Australia (Birch, 1982). However, as our understanding of the P cycle has become more sophisticated, it has become clear that (as always) the overseas experience is not always directly relevant to the Australian environment. Of course, the processes/mechanisms are the same, but the balance and relative importance is not necessarily so, particularly because of the much higher mean turbidity levels in our inland waterways.

It is now agreed that turbid Australian waters can have much higher levels of P in the water column than overseas studies indicate are necessary to initiate excessive algal growth. Hart et al. (1992) proposed that instead of the 10 μg/L used as a guide in the cool temperate northern hemisphere climates

(Vollenweider, 1968), we should adopt a level of 50 μg/L; more recent work now suggests a value up to 100 μg/L (Hart et al., 1993). This is at least in part based on the idea that high levels of turbidity reduce light penetration; they also make less available the soluble orthophosphate in the water column which is necessary for algal growth. At the same time, more attention has been paid to the circumstances surrounding excessive algal growth. The implicit assumption that only external inputs of P are important has been questioned, and it is now becoming more generally accepted that recycling of P from bottom sediments can be a major source of 'bioavailable' phosphate. For example, the paper by Kilham and Kilham (1990) reports a number of examples where the internal loading processes dominate nutrient cycling in lakes and maintain the eutrophic status of these lakes.

It is accepted that there are certain locations in Australia where identifiable point sources of P are able to dominate the water column concentrations (e.g. Hawksbury-Nepean and Murray Rivers, at low flow). However it is almost certain that, at even moderate flows, concentrations are controlled by diffuse sources. From our knowledge of flood delivery, and typical concentrations of P on suspended solids, it is inconceivable that these point sources (sewage, feed-lots) contribute significantly to annual loads. Even in the Hawkesbury-Nepean, it has been estimated that all point sources contribute only around 15-18% to the annual loads (Cuddy et al., 1994).

This may still be an overestimate, because the point source loads are relatively well known, in contrast to the flood loads. It is the universal Australian experience that when floods loads are measured carefully, at any scale, estimates of load are revised upwards. To counter this, it has been suggested that the P derived from sewage/feed-lots is almost entirely bioavailable, in that it is either dissolved or organically bound, in contrast to P from diffuse sources which, it is argued, is dominated by particle-bound P with only very small amounts of P available for algal growth in the water column. Recent work however, suggests that the initial form may be largely irrelevant, because in turbid waters P appears to equilibrate quickly, such that the distribution between forms (e.g. dissolved or exchangeable) is largely independent of origin. Thus we are forced to the conclusion that point sources may be significant in very particular, site-specific, cases, but that in general diffuse sources are the major contributor to loads, and thus to the availability of P in water bodies for excess algal growth.

If we accept that diffuse sources of P are more important than point sources there are both obvious and not so obvious implications for management strategies. Point sources can, at least in principle, be switched off. The effect is immediate. Unfortunately it is not known how long it takes a land management change to impact on waterways. It can be presumed that for larger catchments, involving significant stores of bottom sediment (which may have accumulated over years or decades, in the case of reservoirs), effects will not show for at least years, probably decades, possibly centuries. For instance, it is not clear that the effects of European arrival on sediment delivery in the headwaters of the Darling have yet reached the Murray River. It should also be kept in mind that these time-scales only refer to the reduction in delivery of P to a particular point in a river system, such as a weir or reservoir. It will take further decades or centuries to flush out the P already deposited in such a storage, unless the sediment can be mechanically removed in some way.

The inevitable conclusion from this is that for management to have any rapid effect on water quality, it must be directed at the water body itself, rather than at the catchment. For short-term improvement, there is no alternative. Thus research directed at environmental flow requirements, destratification processes, chemical treatment to remove P, etc, must receive significant support if any improvements are required over the next 10-20 years. This is not to downplay the importance of catchment-based works, which can be carried out for a variety of reasons that benefit the ecosystem, and is essential for long-term reduction of nutrient loads to waterways.

Aims of the Study

The aims of the study are to gain a greater understanding of the limiting nutrient (P) regarded as necessary for blue-green algal bloom formation in inland Australian water systems. This is a relatively large topic and this report does not attempt to be completely comprehensive, but it does provide an overview, through discussion of relevant published papers and work still in progress, which highlights the major issues.

The report concentrates mainly on the movement of diffuse sources of phosphate (P) from non-coastal rural lands. In particular it covers the movement of P from the Murray-Darling Basin (MDB) and other regions of Australia with similar land use and soil type in which, as will be shown later, P moves predominantly with particles. While this represents a large region of Australia, the discussion in this report specifically does not cover the sandy soils, such as found in the Perth region of Western Australia, where P is present in water systems mainly as soluble orthophosphate. Point sources of P, although discussed, are not highlighted in this report. They are by their nature well defined and away from the major cities their input of P to waterways is generally only significant during times of low stream flow. In terms of loads their input is relatively small compared to diffuse loads transported during medium to high flows (Table 1). In contrast, because of the nature of diffuse sources of P, there is relatively little understanding of the sources of P to waterways and we have little information from catchment studies of sediment and nutrient movement through landscapes (i.e. sources and sinks within catchments). This limits development of sound models linking land use, landscape, and nutrient transport.

Table 1. Estimated percentage contributions of different sources to P loads in Australian waterbodies. The Australian climate system which produces these data is discussed later in the report with some further examples. These data suggest that the reported percentage contribution for the MDB needs to be questioned.

location	**Point Sources (%)**	**Diffuse Sources (%)**	**Reference**
Hawksbury-Nepean	15-18 (sewage)	75-82	Cuddy et al. (1994)
Murrumbidgee at Wagga Wagga	8 (sewage)	92 (>95 from gullies & stream banks)	Olley et al. (1995)
Murray at Albury	32	68	Walker & Hillman (1982)
MDB	44	56	GHD (1992)
Lake Burley Griffin	29	71	Cullen & Rosich (1979)
Upper Murrumbidgee	23	77	NCPA (1994)
Drainage to Great Barrier Reef	14	86	Furnas et al. (1994)
Peel Inlet	35	65	Bott (pers. comm.)
Harvey Estuary	15	85	Bott (pers. comm.)
Princess Royal Harbour	75	25	Bott (pers. comm.)
Oyster Harbour	35	65	Bott (pers. comm.)
Swan Estuary	35	65	Bott (pers. comm.)

There are three major sections in this report. The first section, "In-Stream Sources and Bioavailability of P", is used to help define such things as how P is bound in Australia's generally turbid waters, how much P is available for algal growth, and the conditions necessary for algal blooms to occur. These discussions provide the necessary background for the next major sections. In the second major section, "Catchment Sources and Generation Rates of P in Space and Time", the quality of the data available to model the movement of P through landscapes into stream systems is examined, and this section leads logically into the last major section, "The Quest for a General Catchment Model of P Sources".

IN-STREAM SOURCES AND BIOAVAILABILITY OF P

To start this section we first review some of the work carried out by Monash University (Water Quality centre) and the Murray-Darling Freshwater Research Centre in examining the question of the bioavailability of P in Australian river systems (cf. Oliver et al., 1993; Oliver, 1993). Algae can only use soluble phosphate for growth and bioavailable P, in turbid waters, is defined as the amount of P algae can access in the water column (i.e. soluble P and the fraction of P readily desorbable from particles).

Oliver (1993) states that clear water systems have provided the most consistent relationships between the total P (TP) concentration in a water body and algal biomass, and in these cases TP is a reasonable estimate of bioavailable P. In the generally turbid waters found in Australia, however, Oliver noted that this relationship does not hold and suggests bioavailable P is markedly decreased as a result of rapid adsorption onto surfaces of suspended particles. This statement comes from the fact that experimental studies have shown that the initial surface adsorption reactions are rapid (minutes to days), whereas the solid state diffusion reactions have time scales of months to years (e.g. Barrow, 1983; Bolan et al., 1985; Froelich, 1988). In a recent study by Jones et al. (1993), who were examining the assimilative capacity of creeks in Sydney's NW sector (looking at the loading on the Hawksbury River), it was shown that the effect of sewage effluent from the Kellyville STP entering Smalls Creek was that P was largely removed from the water column over a distance of several kilometres.

As a result of exchange processes P can be distributed between a number of compartments in the water column (Fig. 1); (1) orthophosphate ion, (2) aquatic organisms including algae, and (3) bound to mineral and organic particles ranging in size from colloidal to relatively large suspended particles. However, in Australia's turbid river systems it is the rapidity of the adsorption process with particles that limits bioavailable P and must therefore in many cases limit algal growth. The influence of P in turbid waters on algal growth will be determined by such factors as: (1) the rate of orthophosphate replenishment by desorption from particles, (2) the total amount of P available, and (3) stability of favourable environmental conditions (e.g. warm/hot temperatures and still conditions, low flow, stratification, etc.).

Oliver (1993) has used the iron strip method (Van der Zee et al., 1987) to measure desorbable P (DP) in a number of river systems (Table 2) in the MDB and obtained a 1:1 correlation (r^2= 0.99) with bioavailable P in algal assays.

Table 2. Examples of the % desorbable P and amount of orthophosphate ion found in some rivers of the Murray-Darling Basin (Oliver, 1993).

Site	**TP**	**DP**	**%DP**
Darling-Wilcannia	350.7	167	48%
Murray-Albury	84.2	22	26%
Murray-Merein	95.3	29	30%
Ovens-Peechelba	55.0	10	18%
Ovens-Peechelba	49.9	13	26%

Site	Total P[1] µ g/l	Total P[2] >0.2 µ m µ g/l	SRP[3] <0.003µ m µ g/l	Desorbed >0.2µ m µ g/l	Desorbed % >0.2µ m
Murray R.	95.3	74.6 (78%)	2.2	16.5	21%
Ovens R.	49.9	34.6 (69%)	4.7	21.1	60%

1. The total P in the turbid water sample.
2. Total P in the turbid water sample in particles >0.2 µ m.
3. Phosphate in the <0.003µ m fraction is assumed to be soluble orthophosphate.

The results showed:

1. That the %DP using the iron strip varied for a number of river systems in the MDB ranging from 18-83% (Oliver, pers. comm.) confirming that TP is poorly correlated to algal growth in turbid waters.

2. The filtrate from the < 0.003µ m filter had a close relationship with dissolved P. In general, rivers in the MDB had very low P concentrations in this compartment; the exception is the Darling River where the <0.003µm concentration can be as high as 200µg/l (Douglas, 1993).

3. Particles of >0.2mm carried between 70-80% of the TP highlighting the importance of suspended particles as the P source. Work of Douglas (1993)

has shown that, in general, in the MDB the major proportion of P is concentrated in the particle size range 25-0.2µm; again the Darling River is the exception with most of the P concentrated in the more colloid fraction.

Sources of P Causing Algal Blooms

The discussion above focused on some of the P compartments in the water column, but another important compartment is the bottom sediment stores of P (Fig. 1). While the P stores in bottom sediments have been suggested as an important source of bioavailable P under certain conditions (e.g. Lawacz, 1985; Berner and Berner, 1987; Nurnberg, 1987; Caraco et al., 1991), and P retention rates in lakes (e.g. Dillon and Evans, 1993) and the release rates of P from bottom sediments into the water column (e.g. Bostrom et al., 1982; Van der Molen, 1991) have been determined, little is known about the time and/or space relationships of clay-rich (and P-rich) bottom sediments, or the form and transformations of P in bottom sediments in Australa's inland waterways. Harris (1995) makes the comment about Vollenweider models, that they were developed for usually cool, deep and mono- to dimictic Northern Temperate lakes, not for the much warmer, turbid, shallower, generally eutrophic Australian waterbodies. The diagram presented by Harris (1995) for P fluxes in eutrophic aquatic ecosystems (Fig. 2), emphasizes the major role of the internal P load in these systems. In many Australian waterbodies there must be a potential for the internal loading to exceed the capacity of the grazing food chain to assimilate and recycle nutrients, thereby resulting in increases in bioavailable P concentrations in the water and consequently increases in phytoplankton biomass.

Some examples of these stores of sediment-bound P come from studies within the CSIRO Division of Water Resources in Canberra. Sewage discharge from Albury is stored temporarily in a converted billabong on the floodplain of the River Murray. About 1600t of soluble P moves into the bottom sediments of the billabong each year, and only 16t/yr reaches the River Murray. Burrinjuck Reservoir on the Murrumbidge River stores all but about 0.1% of the P which enters the dam. Chaffey Reservoir, near Tamworth, has so much P stored in the bottom sediments that under anoxia this store becomes a source. About 5.5t/yr leaves the reservoir giving a P loss from the reservoir, as a percentage of total available P in the reservoir sediments, of some 6%/year. In spite of this, there is a 6% net gain of sediment-bound P, due to an annual sediment input of about 12%. These examples illustrate only a small part of the complexity of the P cycle in aquatic systems.

We should note the potential for poorly managed storages to become large point sources of P in the future. For example, sediments of either Burrinjuck Dam or the Albury sewage treatment billabong could conceivably become eutrophic, and like the Chaffey Reservoir sediments, export soluble P at rates comparable with current suspended sediment-bound loads in rivers.

It can be seen from these few examples that generally the P loads of potentially resolubilisable P in bottom sediments are very large compared to P loads which can be desorbed from suspended sediments in rivers (cf. Donnelly et al., 1991; Donnelly et al, 1992; Donnelly, 1993). During stratification in reservoirs P release from bottom sediments can occur when oxygen levels fall to around 32% of saturation (Prien and Bernhardt, 1989); can be enhanced as oxygen concentrations fall to lower levels, and can be even further enhanced with increasing sulphate concentration (>10 mgSO_4/l; Caraco et al., 1989). This evidence indicates that this is a store which needs careful management.

The availability to algal growth of P in anoxic bottom water is therefore, of critical importance to the trophic status of the water body. Once anoxia occurs N can be released from the sediments as N_2 gas while P is released as soluble phosphate; that is, major changes in the N:P ratio in the water column can also occur. Importantly from a management point of view, Kilham and Kilham (1990) point out in their study of nutrient cycling in tropical lakes, that the onset of eutrophic conditions can cause the internal load to dominate so that the water body continues to remain eutrophic even when the external load is decreased. Similarly in Chaffey Reservoir, near Tamworth, NSW, internal loads continue to dominate in this eutrophic water body with major algal bloom problems (Caitcheon et al., 1995).

The composition of the bottom sediments in rivers and reservoirs will be a function of the geology, soils and transport processes in the catchment. Phosphate in the clay-rich accumulations of bottom sediments from the Darling-Barwon and Murrumbidge Rivers and Chaffey Reservoir is associated with clay minerals and iron. Bioavailable P can be released from the particles by desorption if orthophosphate is removed from solution (Oliver, 1993), or if redox conditions change in the bottom waters P will be released as ferric iron is changed to the soluble ferrous form. These latter conditions occur during stratification, and it is even possible that in the presence of sulphate (from groundwater and/or sulfate-rich fertiliser input to the water system), or in some natural systems, Fe:P ratios < 2.2 will allow orthophosphate to diffuse into the epilimnion (Tessenow, 1974). Other processes which result in internal P loading reaching surface water are, bioturbation of the anoxic sediments which results in the release of gas bubbles (e.g. denitrification) and entrainment of hypolimnial water into surface water by wind mixing. That is, stratified systems may not need to wait for natural overturn to move at least some proportion of the P-rich bottom water into the photic zone.

Surface soil and subsoil from the Chaffey catchment, near Tamworth, and from the Jerrabomberra catchment, near Canberra, have the same P association with clay minerals and iron (Donnelly, pers. comm.) as found in the suspended sediments worked on by Oliver (1993) and Douglas (1993). In many soils the concentration of P and Fe are higher in the very fine clay fraction (Fig. 3). This phenomenon is well established, so that fractioning the soil into fines has yielded reported enrichment ratios in the range 1.0 to 10 (e.g. Sharpley and Menzel, 1987). If the process(es) of transport from the catchment into the streams, and into a reservoir, concentrate this fine clay fraction then, there will be cases where both non-fertilised surface soils and subsoils will provide relatively P-rich fine clays to a water body.

The Murrumbidge River at the Maude Weir has a suspended sediment concentration of around 40 mg/l during 'normal' river flow. If this amount of sediment has P concentrations similar to that found for subsoils from the Jerrabomberra catchment (av. ~ 0.1% P), and we assume some 25-30% P is desorbable from these particles when they enter a stream then, from this source alone, orthophosphate with a concentration of around 10 μg/l is available for algal growth. Of course if these suspended sediments accumulate as bottom sediments their potential as a source of bioavailable P can be, relatively, extremely large (e.g. Lawacz, 1985; Nurnberg et al., 1987).

Research under this topic should possibly concentrate on selected catchments, then it may be possible to determine the relative bioavailability of P from different sources. Recent research on the Chaffey catchment (Caitcheon et al., 1995) has shown that, although it is a well fertilised catchment, soils derived from Tertiary Basalt in the steep unfertilised upland part of the catchment provides, at least, 90% of the fine sediment reaching the reservoir. The basaltic sediment is P-rich (P_2O_5 values up to 1%; Donnelly, 1993), but in terms of specific sediment yield the catchment is below the average of similar catchments in the region (Fig. 4); erosion is the result of both natural and anthropogenic (tree clearing) processes. That is, while Chaffey Reservoir is the most contaminated reservoir in NSW, with respect to algal blooms, there is evidence that even if a large amount of catchment remedial work was carried out, it would take many years to improve water quality in the reservoir to the point that algal blooms were restricted. Under these circumstances, the short term solution to improve water quality in the reservoir is to prevent resolubilisation of P from the bottom sediments, while catchment remedial work should be explored as a possible long-term solution.

Some Problems Identified

1. The iron strip method (Oliver, 1993) of determining desorbable P in turbid water systems is slow and precision is reduced at low P concentrations (ca. 20-30 μg P/l). It is helping to understand desorption in turbid water systems; how P is bound, and how much P is 'bioavailable' in different river systems. Other techniques which may be alternatives to the iron strip method are, the use of immobilised phosphatase enzymes (Shan et al., 1994), and the use of lanthanum (La), which forms a much more insoluble phosphate compound than Fe (presently being tested by CSIRO Land and Water).

2. There is a problem in not knowing the historical context of our inland water bodies, with respect to algal blooms, suspended P loads, or the ability of bottom sediments in rivers to supply soluble P.

3. The following are poorly understood, regarding bottom sediment stores of P:
   (a) Their spatial extent in Australian water bodies, both in reservoirs and river beds.
   (b) The spatial extent of their ability to function as substrates for algal blooms
   (c) In-stream residence times in river beds. We therefore do not know the time scale on which river-bed stores could be reduced substantially by catchment remedial works, or whether these stores would remain substantial indefinitely, after reducing inputs to levels existing before European settlement. It is not even clear whether the effects of increased sediment delivery, resulting from clearing in the headwaters of the Darling, have yet reached the Murray River.

4. Sediments entering stream systems need not be P-rich, or exclusively top (fertilised) soil, to cause eutrophication in a water body. Even low P soils can resolubilise or desorb bioavailable P in amounts which depending on conditions in the water body, can be used for excess algal growth (e.g. the Murrumbidge River; Olive et al., 1994). All P-containing soils/sediments in a catchment, therefore, should be considered sources of bioavailable P.

Cycling of P in Streams and Storages

To date, the reported Australian studies of P cycling in Australian streams and storages have come mainly from Professor Barry Hart's Water Quality Centre at Monash University in Melbourne. However, the Australian data on the movement of nutrients across the sediment-water interface in freshwater, estuarine and marine environments is sparse. Some research programs at present underway related to this topic are the NRMS funded Darling River study, the CSIRO Multi-Divisional Programs (Coastal Zone and Blue-Green Algal Programs), and the CSIRO Port Phillip Bay environmental study. Programs through the new CRC for Freshwater Ecology (e.g. Chaffey Reservoir) will also provide information on how P cycles in Australian waterways (see AWWA Water J., 22, 1995).

Summary

There is a relatively large number of forms of P in our Australian turbid waters, from inorganic particulates down to small colloids, to organic colloids, through a range of molecular weights, to dissolved orthophosphate. As discussed previously, some forms will dominate more in different environments. For example, in the turbid waters of the rivers in the MDB it appears that P is bound mainly to mineral (clay/Fe) particles (Douglas, 1993), whereas, in biologically active standing water bodies and in subsurface water flow from forested catchments, the organic forms can dominate.

Given the wide variety of processes operating in natural waters (growth, grazing, microbial metabolism, sedimentation) and the rapid turnover of some P pools, it can be argued that over sufficient time almost all the P in the system will be available for accumulation of phytoplankton biomass. What will determine the biomass produced will be the exchange coefficients between all the chemical and biological forms of P in the system and the pool sizes (Harris, 1994).

Clearly, natural P-rich sources of topsoil and/or subsoil, and P-rich fertilised soils, will provide more bioavailable P for algal growth in rivers or reservoirs, than P-poor soils. However, given the ability of clay-rich bottom sediments in these water bodies to store P, under certain conditions, this sedimentary source of P can be potentially the largest source of bioavailable P for algal growth, given the nature of most of our Australian water systems to stratify for some period, at least, annually.

Even to determine the relative bioavailability of P from sewage treatment plants, where P is generally assumed to be readily available for algal growth, is not easy. If the final disposal ponds have significant cell concentrations then substantial quantities of the P will be in biological and organic compartments. As well, if the effluent is discharged into turbid waters, soluble P, and therefore immediate bioavailability, may be removed over relatively short distances. If this is true then diffuse P loads, which are generally much greater than from sewage inputs under medium and high flow conditions, must be given priority in developing management strategies intended to markedly lower P input to stream systems, as opposed to localised "hot spots".

The mechanism of the above rapid removal of soluble P is not clear. Our own data for one case suggest that it is not simple adsorption by the suspended sediment moving independently of the stream bed. It is questionable whether the mechanism is adsorption by bed sediments, without involvement by the suspended load. Further research would be required to show whether the mechanism is adsorption by suspended sediment particles which tend to be removed rapidly from the flow by high exchange with bed sediments. The latter, if found correct, may provide a method of measuring the average velocity of individual particles, compared with the water velocity. This may help to quantify the particle residence time in the system. It should be noted that long residence times imply interaction with large stores of bed sediments.

We acknowledge the contrary findings of Olive and Walker (1982) that the yield of particulate matter from fluvial systems is governed by input of fine soil fractions, rather than in-stream transport processes. This implies short residence times for suspended sediment. Further research may establish that each finding is valid for some landscapes and some spatial scales.

The final conclusions in this section are, therefore: (1) that all sources of P are potentially bioavailable, (2) short term water quality results come from waterbody management, and (3) long term reduction of in-stream sediment stores may come from catchment remedial works, although the feasibility of this strategy is as yet uncertain. Other factors affected by land management changes may provide quick modification of P release rates from bottom sediments, e.g. the loading of organic C (I. Lawrence, pers. comm.), or fertiliser sulfate from the catchment.

CATCHMENT SOURCES AND GENERATION RATES OF P IN SPACE AND TIME

This part of the report examines various issues relating to the movement of diffuse sources of P in catchments, potentially able to be moved into waterways. The proportionate contribution of point and diffuse sources to P loads in Australian waterbodies is illustrated in Table 1. This table was taken from the State of the Environment Report (1996) and shows that, for the twelve catchments examined, diffuse loads of P tend to dominate. The exceptions are where industry and/or urbanisation is particularly intensive and local to the waterbody (e.g. Princess Royal Harbour), and where sandy soils dominate (e.g. Swan Estuary). Although, in the cases of the Hawksbury-Nepean, Swan Estuary, and Peel-Harvey Estuary, diffuse loads still dominate. In view of these conclusions, and particularly in view of where direct measurement techniques were used to determine loads (e.g. Olley et al., 1995), the load estimations by Gutteridge et al. (1992) for the whole of the MDB (Table 1), showing a high point source contribution (44%), seems to be a large over-estimation of its proportionate contribution, in this environment. Later discussion in this section will highlight this point.

It has been established that in environments with soils and land use (agriculture or grazing) similar to those found in the MDB (a large part of Australia), P is mainly associated with the clay minerals, and generally concentrated in the finest particle size fractions (Douglas, 1993). We contend that in Australia the bulk of the sediment-bound P to rivers originates from native soil, rather than from fertilised topsoil. Some eroded soil is not delivered to the streams on a reasonable time scale, i.e. its residence time on the land is infinite for practical purposes. We examine the limitations of knowledge of sediment and P delivery in relation to land use, topography and climate.

A Phosphate Budget

We begin with a spatially explicit P budget, which represents a mass balance for the whole of the land surface, plus the stream network. This will be used as a conceptual tool for considering sources of P in a catchment, for identifying significant gaps in our knowledge of these sources, and for identifying a system whose mass might be more readily balanced in practical models.

A representation of such a budget is as follows:

$$Y + B + FP = S - A + G - F + T + R + E$$

where
- Y = yield at catchment mouth
- B = accumulation in instream bars
- FP = accumulation on floodplains
- S = slope erosion (surface and subsurface)
- A = accumulation on unchannelled slopes (colluvium)
- G = gully erosion

F = fan accumulation
T = erosion loss from roads
R = river channel erosion
E = addition of animal excreta to waterways.

The fluxes are mass/time and are summed over the catchment area. Each source and storage (except E) includes biotic and abiotic components; for example, S includes fragments of organic matter and particles of soil, while FP includes deposition of P in conjunction with sediment and the uptake of P by vegetation.

In principle, an estimate should be available for all of the relevant terms in the above equation for a particular catchment. In very small catchments, R will not occur; in a cropped catchment E will not need to be estimated, and so on. If the source terms are known, total accumulation can be estimated by the difference between the total source fluxes (S + G + T + R + E) and Y. In this case, the uncertainties associated with each of the estimates of source and storage will need to be propagated, and the computed estimate of total accumulation will therefore be accompanied by a large uncertainty. For example, it is so difficult to measure and sum S and A over the whole land surface with accuracy sufficient to be confident that the computed value of S – A has even the correct sign. This is because, in many cases, most of the soil eroded from one part of a slope is deposited lower on the same slope, and does not reach a stream in a reasonable time, e.g. a century. Similar difficulties apply to quantifying G – F.

A model based on such a budget will be as weak as its most uncertain term. This problem will not disappear by improving the model's sophistication. For example, one might systematically vary the number of terms, depending on catchment size and land use. Likewise ,other decisions, such as whether inputs and transfers between storages should be time series or frequency distributions, will not solve the fundamental problem.

Would it be more appropriate to use an alternative budget, that would balance mass just for the stream network, with a term for exchange between the streams and the land? The exchange term may be defined as net stream input, SI = S – A + G – F + T – R – FP. It seems feasible to measure this term directly. Isotopic tracers could be used to partition measured SI into gully and stream bank erosion, on one hand, and surface erosion, with these terms being interpreted in terms of input to streams, rather than processes distributed all over the land. However, it is not necessarily practical to base a model on even this budget. This is because of the difficulty of mapping the in-stream storage, (B etc).

Other information is available or obtainable, and the data might legitimately be modelled, but not in the context of a P budget. For example, stratigraphic analysis can provide valuable information on time trends of in-stream storage, without yielding the spatial information required for an accurate budget. Also, information may be obtained by studying yield directly, without compiling a budget. For example, the slope of the regression equation relating mean annual fluxes (of sediment and P) to a wide range of catchment areas (Fig. 4) is usually < 1.0. This slope indicates one or more of the following possibilities, as scale increases: higher net sediment input to streams (on an areal basis) from the uplands, temporal increase in in-stream storage on the lowlands, residence time increasing. The first of the above possibilities is necessarily true, from physical first principles. Thus consideration of topography, and associated spatial variability of erosional/depositional status, would suggest relatively high net input to upland streams, with negative net input on floodplains. This in no way excludes the other possibilities.

Direct studies of yield may establish correlations between P yield and likely independent variables, like land use type, slope, drainage density. These correlations may be high only over a narrow range of catchment areas, because of the above mentioned variability of erosion and storage with scale. So care is required in combining the results of heterogeneous studies at different (or unknown) scales to compare the consequences of a range of land management options.

In the following, we will first review the quality of the literature presently available on nutrient exports as they relate to different rural land uses, and where in the landscape the export data were obtained. Further, we will review knowledge about each of the terms of the budget equation. The severe limitations of current knowledge will become clearer as we work through each term.

Phosphate Transport and Algal Blooms

Increased monitoring of surface waters in the 1970's revealed the major contribution of diffuse P loads to the total P load exported from catchments (Chittleborough, 1983). Given the link between sediment and P in catchments like the MDB, it may be useful in a review of the quality of the nutrient export data to first review some of the sediment export data, as it relates to different land use.

1. Sediment transport

McElroy et al. (1975) investigated the relative ranking of erosion from different land uses in the USA (Table 3) showing that cropland produces the most sediment. If this sediment gets to streams, croplands are the major contributors.

Table 3. Relative amount of erosion from various land uses in the USA (McElroy et al., 1975).

Commercial forests	1
Mines	2
Construction	6
Harvested forests	11
Grassland	11
Cropland	168

A number of studies by Cornish, from the Forestry Commission, NSW, have indicated that sediment movement from unlogged and logged forests in eastern NSW is generally low (e.g. Cornish, 1989), basically confirming the results reported by McElroy et al. (1975).

These data in Table 3 suggest that there is a very clear distinction of 'surface' soil erosion between the different land use categories. In the following it will be suggested that: (1) surface soil may not always dominate the suspended sediment loads exiting our Australian catchments, (2) that sediment exports for different land use may not always be so clearly defined, and (3) this blurring of the effect of different land use also applies to P exports.

Table 4. Mean annual specific sediment yield (± standard error) from catchments < 10 km^2 on the Southern Australian Tablelands.

Catchment Class	n	Sediment Yield (kg/ha/yr)
Native forest	5	4 ±2
Native pasture	17	19 ±5
Native pasture-discontinuous gullies	5	31 ±2
Cropped	4	57 ±15
Over grazed pasture	2	68 ±19
Pine plantation	7	84 ±42
Established urban industrial	1	160
Native pasture-continuous gullies	1	161 ±68

An Australian example illustrating that the effects of land use on sediment exports can be rather blurred comes from the study of annual specific sediment yield from catchments <10 km^2 on the Southern Tablelands (Neil and Fogarty, 1991; Lawerence, pers. comm.). Although erosion is linked to land clearing and use, it can be seen in Table 4 that the presence of gullies has a major effect on erosion, irrespective of land use.

2. Sediment and Fertiliser P Transport

McLaughlin et al. (1992) discuss the operation of the P cycle in Australia. They note the focus of present discussions on the impact of agriculture (fertiliser application) on eutrophication in waterways and make the point that, unlike northern hemisphere countries Australia's problems have been caused not the intensity of farming but by its extensiveness. Fluxes of nutrients per unit area in Australian agricultural systems are extremely low by world standards and, with a relatively hot and dry climate, the major pathways of nutrient loss are fundamentally different from those in most other countries.

These authors report that the consumption of fertiliser P in Australia has been increasing since 1951, with crops (and others) remaining relatively steady since the middle sixties (~200kt/yr. elemental P), but with a marked decline in the use of fertiliser P on pastures after around 1975, to around 125kt/year of elemental P. The three major uses of fertiliser P for the year 1987 were, pastures (52%), wheat (24%), and barley/oats (10%), and it was estimated by McLaughlin et al. (1992) that the ratio of P input to soils, to export of P in produce, is around six to one. That is, a substantial fraction of the fertiliser P applied to soils is left in the soil.

Garman (1983) combined land use figures with available P export data to show the rate of loss of P from soil under various land uses (Table 5).

With respect to these data McLaughlin et al. (1992) make the following points: (1) that the P loss figure is very high for crops when it is considered that although soil loss is usually higher from cropping soils than from pasture systems, fertiliser placement in cropping soils is almost always below the surface, while in pasture systems there is a marked concentration of P at the soil surface, and (2) the P exports were extrapolated from small plots to a regional basis and do not take into account P storage within the landscape.

Table 5. Rates of P loss for various land uses.

Land Use	P loss (kg/ha/yr)
Forest/conservation areas	0.01
Pasture classes 4&5	0.1
Pasture classes 3	0.2
Pasture classes 1&2	0.3
Crops	3.0
Urban	0.6

In the following we examine the P export data base, with respect to the Australian environment, to judge the quality of the data being used to infer nutrient fluxes in Australian waterways, and also the question of whether top soil dominates the suspended sediment loads in waterways.

American data, from a nation-wide study of 928 catchments (Omernik, 1976,77; Browne and Grizzard, 1979), has shown that: (1) there is an exponential increase in mean TP concentration in streams leaving catchments with increase in the amount of agriculture + urbanisation within the catchment, and (2) there is a significant correlation between stream nutrient concentration and general land use. With respect to this latter point only relatively broad relationships were established between events in lakes and conditions in their drainage basins (e.g. Kira, 1991). An example of the data used by Browne and Grizzard (1979) to establish the existence of the nutrient/land use correlation is shown in Table 6. Correlation coefficients are usually fairly poor.

Table 6. Yield of P from forested and agricultural catchments.

Land Use	Specific Land Use	P Yield (kg/ha/yr)
Agriculture	cropland (general)	0.2 - 4.6
	cropland (corn)	1.6
	improved pasture	0.1 - 0.5
Fostery	general	0.2 - 0.7
	pine forest	0.2 - 0.4
	woodland	0.1

Chittleborough (1983) used data from streams with catchments with similar gradients and soil type, entering the Onkaparinga River in the Mount Lofty Ranges, SA, to show that land use affects sediment and P exports. Chittleborough (1983) concluded sediment and P export is strongly linked to stream flow (rainfall and runoff) and that of the three catchments studied intensive horticulture yielded much greater amounts of sediment and P than the urbanised catchment, which in turn was greater than the virgin catchment.

The pattern of rural land use and its effect on P exports is highlighted in Figure 5, which was taken from the Healthy Rivers Report (1992). The authors of this report suggest that in Australia P exports from forested catchments are low, but clearing such catchments is likely to increase soil erosion. They also indicate that because of the use of fertilisers in agricultural areas P is accumulating in agriculture soils at an average rate of 289 kt/yr and that fertiliser inputs to the P cycle (Fig. 1) are now larger than any other flux. Obviously, any catchment remedial work should aim to decrease the chances of these P-rich soils reaching stream systems.

Soil structural decline is also recognised as a major form of land degradation in the MDB (e.g. Graham, 1989) and this decline in soil structure can have an important influence on the movement of topsoil. The mechanism for the process of increased soil erosion rates and sediment delivery ratios, as a result of the decline in soil structure, is discussed by Hairsine et al. (1992). Briefly soil detached from a structurally-degraded source has finer sediment with slower settling velocities. This exacerbates plot erosion rates and reduces the trapping ability of sediment stores such as fans and wetlands. While there has been no long term monitoring of this effect, it seems likely that soil structural decline has indirectly increased soil-sorbed nutrient transport to streams.

3. Sediment and Non-Fertiliser P Transport

However, it may be too simplistic to suggest that it is only the nutrient-rich (fertilised) top soil which is responsible for the 'recent' increase in the frequency of algal blooms in rural Australia. In the following, a number of examples are briefly discussed which show that other sources of P-containing sediments can cause algal bloom problems.

The first example comes from work carried out by Caitcheon et al. (1995) on the well fertilised Chaffey catchment, near Tamworth, NSW, which contains a reservoir with almost continuous blue-green algal bloom problems. These studies have shown that the major source of P to the reservoir however, comes from the unfertilised, but P-rich, soils developed from upland Tertiary Basalts.

The next example comes from a study of the catchment of the Murrumbidge River; the river has algal bloom problems particularly in the weirs (e.g. Maude Weir). Studies (e.g. Murray et al., 1993) have shown that sediment enters the river from the upper part of the catchment and that, at least, a large fraction of this sediment is relatively P-poor subsoil. Analyses of suspended sediment in the river, under a range of flow conditions, confirm subsoil P values, as do analysis of cores from depositing sites along the river; these cores show no changes in P concentration through time (~40yrs.) which would be expected if fertiliser P was the major source of P to the river.

The next example comes from studies of Leslie Reservoir (the town water supply for Warwick in southern Queeensland) and its catchment (Queensland Water Quality Report, 1992). The catchment can be regarded as pristine in the sence that there are no obvious sources of P in the catchment. However, the reservoir has algal bloom problems (a well publised closure of this storage occurred in March to July 1992) and water quality monitoring has shown that, at times, the reservoir can show relatively high internal P loadings (QDPI water quality data).

The final example comes from studies examining the relationship between rainfall and nutrient export coefficients in South Pine catchment, southern Queensland (Cosser, 1989). Stormflow was the major determinant of nutrient export and coefficients had to be based on kg/ha/mm (rainfall). Cosser (1989) found that 75-89% of the P loads to the river were carried in 2.8% of the time and significantly, even with what are regarded as P-poor soils, P concentrations in runoff were up to 2.48mg/l. If runoff is a significant proportion of the water in the system after storms then these concentrations are high enough to produce significant algal growth.

4. Extreme Events, P Exports and Algal Blooms

The major nutrient loadings to Australian rivers (and weir pools) and storages occur during storm events. It has been noted that storm events can have impacts on phytoplankton dynamics in a lake for years after the event (e.g. Ferris and Tyler, 1992). The increased nutrient loadings to water systems can affect the ecological structure in the system, or if the nutrient loadings exceed the capacity of the grazing food chain's ability to assimilate and recycle nutrients, then blooms can occur. Thus extreme events, with river regulation, can have a major effect on the overall trophic state of water bodies.

It has been suggested that the role of extreme events in Australia on sediment and P exports to rivers and storages and their effect on algal blooms has not yet been fully evaluated (Harris, 1994). Some examples of the importance of storm events in transporting sediment and nutrients are described below.

Studies have shown that essentially all sediment entering the Murrumbidge River enters from the upper part of the catchment (Murray et al., 1993). The large flood events which have occurred in the Murrumbidge River catchment over the last approximately 40 years have been responsible for around 35% of the total sediment load transported past Wagga Wagga in NSW (Olive et al., 1994). Two of these large floods (1 in a 100 yr. events?) in 1956 and 1974 were responsible for nearly 50% of these large flood loads, and sediment transport occurred over a very short time. Smaller flood events account for most of the 65% remaining load transported over this period. Translated into annual sediment and P loads from this approximately 40 year record, some 600,000 tonnes of suspended sediment moves past Wagga Wagga annually, and from P concentration data through several floods, with varying sediment sources but similar low-P values (~0.1%P), on average, ~600 tonnes of P associated with the sediment pass Wagga Wagga annually (Olive et al., 1994).

Chittleborough (1983) provides another example of the event driven nature of sediment and P transport in Australia in his Mount Lofty Ranges study. It was shown in this study that >50% of the annual sediment loss and 54% of the P loss from one catchment occurred during two storms representing only 8% of the annual flow. Cullen (1995) mentions the Monkey Creek data in an article in this journal and comments that a storm event caused 61% of the P to be moved in 1% of the time of this one year study. The weather in Australia produces many storm events and a number of Australian studies have now shown that the major sediment and P loads are generally transported over short time intervals, as a result of these storm events (Cullen et al., 1978; Hart, 1983; Edwards, 1987; Cosser, 1989; Gutteridge et al., 1992), however, the link with excess algal growth is not always clear.

As mentioned in the opening to this section extreme events can affect the ecological structure within a water body and the P loading may be incidental. For example, in the estuarine environment of Oyster Harbour in Albany, WA, massive areas of seagrass have been lost and macroalgae has proliferated in relatively recent times (Mills, 1987). The major land development which occurred after European settlement in eastern Australia occurred only relatively recently in southwestern Western Australia. It is possible that the initial flushes of sediment following land clearing in the Kalgan catchment, which terminates at Oyster Harbour, provided a blanket over the seagrass destroying large sections and providing an ecological niche for the macroalgae to proliferate. It can be speculated that similarly the relatively recent loss of, at least, some of the macrophytes which appear to have dissapeared from the MDB rivers may be linked to a few extreme events, thus providing an ecological niche for carp and the proliferation of blue-greens.

5. Flow, P Loads and Algal blooms

Jones (1994) has suggested that background nutrient levels in parts of the Murray-Darling system are high enough to cause algal bloom problems, because even a relatively small biomass of cyanobacteria, if toxins are produced, can cause severe problems in calm storages.

Previously evidence was presented that P-poor subsoil can be a major input to water systems and that even P-poor soils can provide sufficient bioavailable P for excess algal growth. It is further suggested that P loads to large river systems may have not changed significantly over the last few hundred years. This evidence comes from the CSIRO Murrumbidge River study which indicates that: (1) a large fraction of the sediment entering the river is subsoil (Murray et al., 1993), (2) suspended sediment in the river has P values (~0.1%P) similar to subsoil values noted around Canberra, and (3) cores taken from sites of deposition along the river show no change in P concentration over an approximately 40 year period (i.e. no expression of the marked changes in fertiliser use over this time; McLaughlin et al., 1992).

It was suggested by Jones (1994), that because buoyant cyanobacteria are favoured by stable water column conditions, a major management technique for preventing cyanobacterial blooms should be artificial destratification and river flow management. These strategies appear to be powerful tools for short-term reduction of P release from bottom sediments and/or markedly decreasing the time that the cyanobacteria spend in the epilmnion, in order to improve water quality. As implied earlier, it may be appropriate to supplement these strategies with land management aimed at reducing inputs of organic carbon and sulphate to the rivers, to further reduce P release. The above strategies may or may not be appropriate for long-term management also. The uncertainty is because we do not yet know the relative importance of P release from bottom sediments, on the one hand, and delivery and in-stream transport of sediment-bound P, on the other (see *Problems Identified* in the previous section).

6. The Quality of the P Export Data

In the discussions above, referring specifically to the MDB, and basins with similar land use and soil type, it has been shown that land use does affect soil and nutrient transport. However, it was also shown that subsoils can be a major component of the transported soils and even P-poor soils can provide relatively large amounts of bioavailable P to water bodies for excess algal growth. The statement by Harris (1994) that "in general it seems that, in Australia, phytoplankton dynamics are dominated by irregular and infrequent extreme events" appears to be true and highlights again the importance of diffuse P loads to water ways. Harris (1994) qualifies this statement by adding that the *long term studies* necessary to prove this strong link between climate variability and algal blooms are few.

In the following we first address the question of the quality of the P export data presently available for use in models estimating nutrient export to streams based on land use, rainfall, soil type, etc. Following this we will then examine what parts of the P budget, proposed at the beginning of this chapter, have flux data and how limited is the data base necessary for determining a P budget at this time. It will be shown in this section that given the complexity of nutrient transport processes in rural Australia and the quality and quantity of the presently available P export data, it is obvious that the data is lacking and that more studies such as that carried out by Cosser (1989) in the South Pine catchment in Queensland, or, for example, the QDPI Darling Downs runoff and erosion studies (Freebairn and Wockner, 1986 a&b), combined with nutrient studies, would help build up this presently deficient data base.

Some problems with the existing data base are highlighted below:

1. The hydrology of Australia shows the most temporal variability in the world (Finlayson and McMahon, 1988). There are many examples showing very large annual P exports for a small fraction (<10%) of the annual flow. As well, as catchment size increases there is a greater potential for the spatial distribution of erosion to differ for different storm events. Wetter regions of the catchment are more susceptible to loss of sediment (although any disturbance to the land will increase the potential of increased loads of sediments and P to streams). Therefore, P exports are highly variable in space and time. Where P export data are available, the range of values reported is usually very large.

2. The general lack of data to estimate loads, which means, at times, using non-Australian data, which may not be applicable to Australia,

and having to rely on a few "key" Australian studies where we cannot afford to question the reliability of the study and its applicability across a range of different environments (e.g. climate/soil/topography/vegetation). The role of topography will be discussed later.

3. P export data comes from plots, and from the ends of catchments. There are no studies of P fluxes through catchments. Clearly what comes off a plot may not reach a stream, and what comes from the end of a catchment is the result of many processes not understood or quantified.

Yield data from the SE region of the MDB (Fig. 4) suggests that P yield increases perhaps a little more slowly than sediment yield, as catchment size increases. Sediment becomes finer as scale increases, and there are higher P concentrations in finer soil fractions. The data then more convincingly suggest that P yield increases more slowly with scale than does sediment yield. This suggests that P is delayed relative to sediment in transport, i.e. that P has longer residence times relative to sediment, as scale increases. This conclusion is supported by our preliminary investigation of P/Fe ratios, suggesting that the iron oxide/phosphate coatings on fine particles are abraded during transport in streams.

Both changes with increasing scale, *viz*. finer suspended sediment and loss of P, suggest that sediment yield is not in equilibrium with the climate and the present state of the landscape. With a catchment scale of millions of square kilometres, for example, the exported sediment does not include silt and fine sand fractions, so that yield of these fractions is not in equilibrium. Further, loss of iron oxide coatings, so that this fine but heavy fraction does not reach the outlet, confirms that P yield is not in equilibrium with the land at this scale. Lack of equilibrium means that we cannot interpret P yield, for example, in terms of topography and current land use. Another way to state this is that, for yield of some fraction, residence times of material in transit are too long for yield to reflect the current state of the land. At sufficiently small catchment scales, yield should be in equilibrium with the land, that is, residence times should be less than the time since the current land use began. However, we do not yet know what these space and time scales are.

Clearly land use affects P exports to rivers, and agricultural-dominated catchments provide greater loads of soil and P to rivers compared to forested catchments (e.g. Omernik, 1976 & 77; Brown and Grizzard, 1979; Healthy Rivers Report, 1992). But our present data base and empirical models are inadequate to describe the combined effects of erosion and transport of P. In fact, our models do not adequately recognise such important factors as topography, or in-stream sediment storages and their dynamics.

Terms of the P Budget

In this section we look at the individual elements of the P budget and use the nutrient export data discussed previously, with some actual examples, to highlight the difficulty of presently determining a P budget:

Yield (Y) Listed below are some examples of studies where stream loads were used to estimate exports:

1. Catchments with various land uses in the Mt Lofty Ranges of South Australia (Buckney, 1979; Clark, 1988; Chittleborough, 1983).

2. Thirty-six non-point source catchments from the south-west region of Western Australia (summarised by Bott, 1993).

3. The catchment of the Peel Harvey estuary, WA (Birch, 1982; Birch et al., 1986)

4. Catchments in the ACT region (Cullen et al., 1978; Cullen and Rosich, 1979).

5. The catchment of the Gippsland Lakes, Victoria (Bek and Bruton, 1979).

6. Some Melbourne, Victoria, catchments (Campbell, 1978; Kesari and Vass; 1982)

All of these data, by definition, come from measured loads in channels and are first order functions of catchment area.

Slope Erosion (S) There are some data for surface soil movement from erosion plots on hillslopes, for example:

1. Soil and nutrient loss from grazed improved pasture near Canberra (Costin, 1980).

2. Hairsine et al. (1993) provide field scale data of sediment and N and P movement for two extreme events for a cropping situation near Cowra in NSW.

3. There is a recent study of nutrient movement within small cultivated and pasture plots in the Onkaparinga Catchment (Chittleborough, pers. comm.).

4. McColl (1979) and Sharpley and Syers (1979) reported exports of P from grazed and ungrazed pastures in New Zealand, and showed that exports increased after adding fertiliser to the pasture.

5. Data showing that nutrient exports are generally higher from cropped lands comes mainly from American studies (e.g. Menzen et al., 1978; Alberts et al., 1978).

These data show a correlation with land use, but with a large coefficient of variation. Chittleborough (pers. comm.) has reported some subsurface data for a small plot in the Onkaparinga catchment in South Australia, but there are no data on how far the nutrients move in the landscape and if they could reach a stream. New data could be generated by measuring the P content in the soils of erosion plots for which there are many years of erosion data in parts of Australia. For example:

1) The NSW Soil Conservation Service's long term plot studies on cropping management practice on runoff and soil loss (Edwards, 1987; Packer et al., 1992) which are long term studies at a range of localities in the state.

2) The QDPI field studies of runoff and erosion on the Darling Downs (Freebairn and Boughton, 1985; Freebairn and Wockner, 1986 a&b) which is also a long term study at a variety of scales for the predominately heavy textured soils of this region.

<u>Gully Erosion (G)</u> There are no export data currently available to test the hypothesis that gully walls are a significant source of P. Samples from a large gully near Canberra, Jerrabomberra Creek, show the same mineral-P association (Donnelly, pers. comm.) as suspended sediments in many of the major rivers of the MDB (Douglas, 1993) and, as mentioned in previous sections of this report, can contain enough P to be an important source of bioavailable P for algal growth.

<u>River Channel Erosion (R)</u> – no data

<u>Animal Excreta (E)</u> – in the rural environment not involving intense farming industries-no data

<u>Erosion losses from roads (T)</u> – no data.

<u>Accumulation (B, FP, A, F)</u> – again no data.

A Gross P Budget: Variability in Space

In catchments within the MDB gross P budgets, similar to that developed as an Australia-wide P-budget by McLaughlin et al. (1992), could be developed using the mineral-P association. The export of sediment, on a per unit area basis, decreases as catchment area increases. This follows because the slope of the sediment plot (total yield basis) in Figure 3 is less than one. The pattern of change with increasing scale is explained in part by change in catchment form. A portion of sediment moved by sheet erosion will be stored in footslopes and on floodplains adjacent to hill slopes; with increase in catchment area the products of sheet erosion are increasingly trapped, due largely to reduction of slope and contour curvature. Drainage density decreases with increasing catchment area, indicating that the average flow path to a channel increases with area. Larger catchments are therefore more likely to trap particles than smaller, generally steeper catchments. In fact, at large scale there are floodplains, which are depositional structures formed by net removal of suspended sediment from the streams to the land. Floodplains and in-stream sediment stores (e.g. bars, streambed deposits), complicate the gross budget and yield estimates, as it is difficult to quantify the rates at which these deposits are increasing (or decreasing) currently.

Statistical Relationships

Empirical models using statistical relationships (such as decision support models like CMSS) rely on the use of correlations between, for example, yield (Y) and land use, between P and rainfall intensity on plots, etc. The influence of these factors on nutrient yield have not been adequately established either through empirical relationships, or from specific studies in the catchment of interest. The Chaffey and Murrumbidgee examples illustrates that land use is not always the controlling factor. The importance of proximity to streams, slope, soil type and hydrological regimes have only partially been incorporated in such models. No consideration appears to have been given to going further, and explicitly recognising either the depositional nature of floodplains, or the importance of stream banks as sediment sources, in modelling sediment and nutrient exports. Therefore they cannot be used to predict the effects of land use changes on downstream water quality.

Summary

The discussions above on P exports from rural Australia refer specifically to the MDB and similar basins with the same land use and soil type. It was shown that land use does affect soil and nutrient transport to streams. However, it was also shown that subsoils can be a major component of the transported soils and even P-poor soils can provide relatively large amounts of bioavailable P to water bodies for excess algal growth under the right conditions. The statement that in Australia phytoplankton dynamics could be, to some degree, dominated by irregular and infrequent extreme events, if true, highlights again the importance of diffuse P loads to our inland rivers and storages and algal blooms. If we accept that the frequency of algal blooms has increased in our inland waterways then it is possible that this is, to a large degree, linked to the most significant change we caused in these waterways (i.e. the restriction of flow by storages and weirs).

With respect to the P budget, some yield (Y) data exists but there is little idea of any meaningful relationship with land use, etc. The mechanisms, magnitudes and processes of P transport in catchments are poorly understood. At present little or no information is available on the relative importance to surface waters of various sediment and nutrient sources. There is also insufficient knowledge of the behaviour of P in streams and rivers. Such a 'P Budget' as presented in this section is obviously unrealistic at present and, as a concept, may also be unrealistic because catchment complexity may preclude adequate application of this model approach (see later discussion).

Potential models and model development to give the best approach to understanding how P moves through large catchments and into water systems, and along river channels, are discussed in the next major section of this document. However, the model must be based on a good insight into the functioning of the system as a whole. It is this insight which allows the modeller to select between alternative models. The insight can come from accumulated experience, either first hand or from the work of others; but it is also necessary to obtain real information on the particular system under consideration, though this may take many forms (e.g., information on climate, soils, relief, etc.).

A conceptual model approach has been used by Jakeman and co-workers (e.g. Jakeman et al., 1991) using an extended version of IHACRES (see later discussion) for both regional prediction of hydrographs, and predicting the effect of climate change on catchment runoff. The major insight necessary for this type of modelling is in the choice of model structure, or the class of model structures considered. Through clever choice of the model structure, explanation of 80%-90% of the variance is obtained routinely with few parameters even in circumstances where nonlinearities apparent in the data would defeat regression-type modelling.

This 'System Identification Approach' (Fig. 6), is presently being used in a combined CSIRO/ CRES-ANU study of P transport in the Namoi Basin (NRMS Project Nos. D5010/R5061). Some of the advantages of this model, with respect to the discussions above are that: (1) it uses physical landscape parameters to develop meaningful relationships with land use (on a daily basis) and annual P export to streams, (2) the 'key' processes can be identified and base data inputs need not be very large, and (3) the model can be applied to large to medium scale catchments, but predictions from the model can also be at the paddock scale. A further advantage of this modelling approach is that it involves modelling the flow regime along with sediment/P dynamics. This would avoid the need to prejudge the relative roles of flow and P transport in determining whether algal blooms develop.

THE QUEST FOR A GENERAL CATCHMENT MODEL OF P SOURCES

Introduction

The preceding discussion makes it clear that the availability of P in waterways is due to the interaction of a large number of physical/chemical/biological processes. Few of these processes have been well characterised individually, even at the finest scale, and interactions may be both subtle and difficult to observe. In contrast, what data can be obtained on P concentrations and fluxes in stream networks is seldom of sufficiently high quality to suggest an information content equivalent to more than a few degrees of freedom. Consequently, any model developed to describe the P response of these systems must either be relatively simple, and unlikely to describe the subtleties of behaviour which are sometimes inferred; or require additional, largely unsupported, asssumptions (i.e. prejudices) to evaluate the extra parameters required by a more complex model.

Because water takes a primary role in the movement of P, whether in soluble form or attached to particulates, and is often also a determining factor in P generation, models of P transport must implicitly include a hydrology component. Models for P delivery to water courses cannot therefore do better than the best water yield models, and in practice must be expected to be somewhat less accurate in their predictions and representations of catchment behaviour. For this reason, it is appropriate for catchment models for P to begin with a successful water yield model; though it is highly unlikely that a water yield model can become a useful P model by simply tacking on a P transport component.

However, the extension to a P yield model is not as simple as is sometimes assumed, due to a number of subtleties. Firstly, although predicting losses due to evapotranspiration is the key to any successful catchment water yield model, there is a highly non-linear relationship between such losses and precipitation. The difficulties are compounded for solutes because mixing processes must now be taken into account, and with varying concentrations it now matters precisely where the losses occur. Secondly, for particle-bound P from surface and gully erosion, the overland flow component of the hydrograph is most important. But the magnitude of this component is also a highly non-linear function of precipitation and location, depending on a delicate balance between precipitation rate and history, soil and geomorphic properties, and topography.

Besides combining hydrograph models with particulate or solute transport models, it is also necessary to characterise the relationship between P concentration and particle size, and chemical transformations such as adsorption/desorption phenomena, as the P moves through the system. The magnitude of these difficulties does not appear to have been well recognised at present, particularly with event-based models where extensions for solute and particulate generation and transport appear to be simply "tacked on", with little attempt at independent calibration. An absence of high quality, high frequency catchment scale data has resulted in poorly calibrated and often inappropriate models.

The challenge for the modeller is to develop a (generally quantitative) description of the system under examination which both represents the actual behaviour of the system and suggests an explanation in terms of known processes; uses observable data (appropriate to the scale of the system); and is capable of discriminating between alternative management strategies at an appropriate scale. The term "general" in the title of this section implies an approach which is applicable to the range of spatial and temporal scales, as well as the breadth of concerns, relevant to catchment management. An additional requirement is sometimes considered, that the model is also sufficiently simple to be useful directly to managers, without further intervention by a modeller beyond the development phase. These simultaneous constraints restrict significantly the range of models which can be developed; in practice, several constraints are ignored to give a partial solution to the above challenge, while results from more than one of these partial solutions may be combined in search of a more comprehensive understanding of the complete problem.

Definition of a "model"

The term "model" usually refers to a mathematical representation of the relationships inferred between different properties of a system, allowing quantitative inferences about aspects of the system behaviour. Such a definition is too restrictive for our purposes here, as it obscures the connection between the mathematics or computational scheme (which is merely an inferencing tool) and the more profound relationships of which the mathematics is a representation. For our purpose, a model is an insight into the behaviour of a system, its processes and interactions. It necessarily involves simplification, and a judgement about the dominant processes and interactions responsible for the major features of the observed behaviour of the system. Put in this way, it is clear that the development of a model is a subjective process. It is also clear that any form of engineering activity, environmental manipulation or even the simplest form of data collection involves the use of some form of model.

Current Catchment Models

Many models of catchment transport processes which can be used to simulate P transport are detailed in the book by Ghadiri and Rose (1992), and a few others are mentioned below (e.g. CMSS, AEAM). The reader is referred to Ghadiri and Rose (1992) for the meaning of most model acronyms. The Ghadiri and Rose book discusses many of the assumptions and limitations of these models, selected from an Australian viewpoint, in far greater detail than is possible here, and the reader is referred to the book for further details. Many of the models discussed give reasonable descriptions of P transport, given the system (mostly small-scale) and/or viewpoint for which the models were originally developed. The developers frequently lay claim to some degree of generality, but none of these claims will bear close inspection. In fact the very diversity of the models vindicates the standpoint taken here, that catchment complexity precludes the possibility of such a general model.

Model Classification

No satisfactory general catchment model for P generation and transport exists at present. There are a number of models which have been used for P with greater or lesser degrees of success, but all have limited scope. Most event-based models are adaptations of hydrological models which have been extended to nutrient and particulate transport. Consequently, calibration and testing has been overwhelmingly directed towards hydrograph prediction, with very little effort involved in formulating or justifying the structure of the extensions. Extension of hydrographic models to particulates and solutes requires additional assumptions. For instance, introduction of solutes into systems of storages requires mixing assumptions; while particulate transport generally requires assumptions to be made about particle detachment, and the coupling between water flows and particulate movement. Modelling of particulate borne nutrients such as P requires still further assumptions, involving enrichment factors and adsorption/desorption behaviour. The analysis here is therefore more of generic types than of particular models, with a view to determining their potential for yielding the desired general model.

Available quantitative catchment models which can be used for P can be classified in terms of the constraints which they attempt to satisfy. One system used in general hydrology is to divide the class into empirical models, physical models and conceptual models. We begin by using this classification, while recognising that no clear distinction exists between the elements of this classification. However, we shall argue that the term "physical model" when used without qualification is misleading. A further category, which we call 'model frameworks', and which does not fit the above classification has recently begun to emerge, and we consider this category separately.

1. Empirical models

In its purest form, this type of approach is used to parameterise the yield of a material (e.g. P) in terms of a number of measured variables which are assumed relevant. Relationships between observed system inputs and outputs have been used extensively in hydrology, as in other branches of science. Given a sufficiently large data set, it is often possible to obtain statistical relationships between the yield and other variables which explain the yield variance to a very high degree. For environmental conditions representative of the original data set, such models can give highly accurate predictions. General data requirements are high, as it is necessary to cover the range of potential conditions as completely as possible, but only relatively few variables may need to be measured.

These relationships are almost invariably linear, or quasi-linear (cf. power law relationships) because of the difficulty in parameter estimation otherwise. The subjective part of this type of modelling is in the choice of the regression variables, and this represents the main insight gained from the data. There is however, no guarantee that the established relationship is in any sense causal, or that the variables are independent. The approach is only possible if the system response is stationary, depends on a few variables at most, and/or is approximately linear (or linearisable cf. USLE). In some circumstances it may be possible to reduce non-linearities, either by averaging over time (e.g. mean annual yield, cf USLE; CMSS, Davis et al., 1991a), but this means that the model is only adequate for representing the dynamic response of the system over periods large with respect to the averaging time; or by the use of a non-linear pre-filter for the input data (e.g. the use of "effective rainfall" in the classical Unit Hydrograph approach for predicting a catchment hydrograph from measured rainfall).

A Unit Sediment-graph (by analogy with the Unit Hydrograph) has occasionally been suggested for modelling sediment production, but this approach has only recently been attempted for dynamic modelling of the P yield of catchments (Raghuwanshi, et al., 1994), mainly due to the lack of suitable data sets. In principle it could be used to model the P response of a single catchment. While empirical models are capable of representing observed behaviour very well, very little insight is obtained into the causes of the observed relationships, and little guidance is available for management options. Directly varying one of the regression variables to simulate environmental change (management induced or otherwise) is not generally possible, as (for example) parameter estimates may be highly correlated. Implemation of such a model will usually require relatively sophisticated statistical and data handling skills which may not be available to all managers.

2. Physical models

Physical models are an attempt to describe system behaviour through combining descriptions of known physical processes and their interactions obtained at smaller (distance) scales; they represent an application of reductionist philosophy of the type which has permeated the successful development of most sciences over the past few centuries. Physical processes, thought to be relevant from previous experimental and theoretical work at point scale up to hillslope scale, are combined and the system response obtained by integrating contributions from small landscape elements. Subjective judgement is required in selecting the range of processes thought to be important, and their interactions. At best, this is determined by the insight of the modeller into the particular system being modelled, based on experience with other systems; in practice it is often determined by familiarity with, or commitment to, a particular model. In principle, models of this type are capable of predicting catchment response from measured catchment properties (topographic information, hydraulic conductivity, soil type, etc.), and the attractiveness of this goal is one of the main reasons for the recent emphasis on this class of models. In practice, there are a number of major difficulties which need to be overcome, including heavy data requirements, inappropriate model representation, lack of insight, and mis-identification of process.

Detailed process modelling is based principally on analysis of laboratory or plot scale experiments, and this results in a bias towards processes which are important at the small scale. So, for instance, many hydrological models are extended to include solute transport by including a term involving a constant dispersion coefficient, appropriate to data from homogeneous soil columns, whereas field data requires at least a "dispersion coefficient" which increases with scale. Because processes have been characterised experimentally at small (distance) scales, parameters for the model (e.g. hydraulic conductivity, dispersivity, roughness coefficient) are usually measured from small-scale data. Universally, as catchment scale increases, so does heterogeneity, and so do the data requirements for sound parameterisation of physical models. For instance, use of Darcy's law to model water fluxes presupposes that hydraulic conductivity and soil water potential gradients are both well defined and measurable at a scale which is comparable with the spatial resolution of the model. Normally, the largest measurement scale that is achievable, even in principle, is very much smaller. Then, due to nonlinearity of soil hydraulic properties, there is no sound basis for aggregating measurable data to derive "effective" values of these parameters. In fact, consistent "effective" values may not even exist, e.g. the value of soil water potential at the surface that is "effective" for modelling exchange of soil water between the surface and the atmosphere may differ from the effective value required to model soil-water dynamics. Even ignoring these problems, for larger catchments increasing effort is required to deal with heterogeneity above measurement scale, regardless of the spatial resolution of the model. In a large (heterogeneous) catchment, adequate specification of catchment structure involving both surface and subsurface properties is practically impossible. Using "effective" values of parameters in order to obtain a spatial aggregation at higher than measurement scale requires calibration, effectively nullifying the prime advantages of the physical model.

Because catchment flow processes are complex, involving sometimes subtle interactions between a large number of processes, it is not possible to select a priori which processes and interactions are dominant. This leads to the reductionist nonsense implicit in basing a highly distributed model on the catch-all P budget of section 2.1. Although the representation of individual processes may be very good at the laboratory or small plot scale (cf. Richards' equation, or the advective dispersion equation) with a minimum of parameters requiring estimation, at the catchment scale the combination of even a small number of processes leads to a level of parameterisation which exceeds the information that could be obtained from observations at this scale. This over-parameterisation then requires ancillary information to be obtained (for instance from plot experiments) in order to completely specify the model. Inevitably, this again involves some degree of curve fitting or model "calibration".

In order to reduce the number of parameters which need to be evaluated, all process-based models select a few processes which are modelled in some detail, and effectively ignore the remainder. This subjective identification of the dominant processes constitutes the chief insight of this form of modelling, but also represents the source of its main limitation; because the dominant processes change with scale and circumstance, no single model can be in any sense "universal" or general. In fact, it follows that a model will give a progressively more distorted picture of system behaviour as scale increases further beyond the scale at which the model is valid. For instance, topographically explicit models (such as the TOPOG family) could in principle give a good basis for P modelling in small, steep, headwater catchments where the residence time of water and particulates on the hillslope is relatively large compared to that in the channel network, whereas models such as HSPF or SHE are based on complementary assumptions, and do not have the same level of topographic detail. This lack of universality, and the underlying assumptions, are seldom acknowledged.

Application of these generalised process-based models to data from catchments which were not originally part of the model development introduces a number of dangers. There is the danger of process mis-identification. Because of the high degree of parameterisation inherent in all such models, it is almost certain that a reasonable fit to the response data can be obtained, even if the actual processes are inappropriate for the new catchment. The fit to the catchment behaviour will not generally be as good as that of an empirical model, however, as the predicted behaviour is constrained by the assumed model structure. This danger is exacerbated if the model application is carried out by an inexperienced modeller, as the most important aspect of modelling is the insight into which processes are significant for the particular application, and hence which models may reasonably be considered. The key danger is that it is possible to predict a catchment response with any of these models without any consideration of the actual applicability of the model.

Having highlighted the difficulties with using process-based or physical models, it is necessary to also stress some of the very great advantages of this sort of modelling over other forms. Firstly, because physical modelling is based on relatively well characterised physical laws and processes, the explanatory power of these models is maximised. Unlike empirical models, relationships may be causal (provided the relevant processes have been identified), and system behaviour well outside the range of existing data can sometimes be inferred (cf. climate or land use change). Explicit ackowledgement of model assumptions yields the possibility of understanding the limitations and range of validity of application of the model, and suggests possible extensions to overcome these limitations. This ensures that confidence in the transferability of such a model is maximised.

3. Conceptual models

These models are in some sense in between empirical models and physical models, in that they attempt to represent observed data in terms of quasi-physical storages and interactions, but fall short of an explicit process description. Conceptual models attempt to represent the structure of observed system data using a parametrically parsimonious description. Models are often represented as a configuration of "black boxes", with the relationship between inputs and outputs determined according to some pre-determined algorithm, usually based on some quasi-physical principle, and individual boxes weakly coupled in some conceptual scheme. Examples of this type of model in hydrology are IHACRES (based on the Unit Hydrograph approach, with series/parallel assemblages of linear reservoirs); and the AWBM model (involving series of non-linear storages).

Examples of "box" conceptual models abound, of different degrees of sophistication, and they are particularly useful when information is scarce, as is often the case with environmental systems. As further information becomes available about any part of the system response, the model structure can be progressively refined, and extra parameters introduced as permitted by the increased degrees of freedom inherent in the new information.

Recently a subset of this class of models has shown considerable promise for representing transport in catchments (e.g. IHACRES). Models are developed using a systems identification approach to system-scale data. Model parameters must be calibrated from observed data, but as the model structure is chosen on the basis of observed system behaviour, the parameter space is usually well differentiated and parameter estimation is relatively robust (compared to physical models). In practice, model structure is inferred from a large (though restricted) class of possible structures, which ensures that an optimum structure is identified. The approach is independent of the scale of the system, because model structure is inferred directly from system response data, independent of scale dependent assumptions.

Strictly speaking, this approach suffers from the same drawbacks as the empirical modelling approach, being based on local data. But because of the identification of model structures from the data, a much higher degree of differentiation between different models is possible, and parameter estimates are less likely to be correlated. If model structures are found to be stable across a range of systems, this approach yields a powerful method of comparing their responses. By comparing parameter estimates for the different sytem models, it may be possible to discern regional patterns in parameter values which can be correlated with physical catchment descriptors. Also, because of the quasi-physical nature of the components of the conceptual model, it is possible to vary parameter values to simulate the effect of different management options or changing environmental conditions (eg. greenhouse effects) in a consistent way. Jakeman and co-workers (e.g. Jakeman et al., 1991) have utilised an extended version of IHACRES for both regional prediction of hydrographs, and predicting the effect of climate change on catchment runoff.

The major insight necessary for this type of modelling is in the choice of model structure, or the class of model structures considered. Through clever choice of the model structure, explanation of 80%-90% of the variance is obtained routinely with few parameters even in circumstances where nonlinearities apparent in the data would defeat regression-type modelling. A good conceptual model will reflect well defined features of the data, such as rate or time constants, and associated storage sizes, leading to estimates which do not depend critically on the precise details of the chosen model. However, the actual estimates of these parameters are obtained by calibration, and cannot usually be determined from the conceptual model itself; calculation from "first principles" requires the use of a physical model. Conversely, parameter estimates which are poorly defined or highly correlated are unlikely to be critical, and hint at possible redundancy in the model.

A little thought shows that the "physical" models mentioned above are in fact simply assemblages of conceptual models (e.g. Darcy's law) at a smaller scale. There is in fact no firm distinction possible between these classes of models. What we have called "physical models" are models based on processes characterised at a smaller scale, and found to have some degree of universality. The successful conceptual models at one scale become the physical processes at another; the difficulty experienced in catchment modelling is the lack of robust conceptual models at this larger scale, due, at least in part, to the paucity of relevant data.

For complex systems such as catchments, the above discussion shows that physical models, which attempt to couple many processes together in what is inevitably an ad hoc manner, are unlikely to yield a satisfactory modelling approach in general. The role of conceptual models has been traditionally denigrated by physical modellers, but our discussion suggests that these models, with their emphasis on system identification through the analysis of catchment response data, at least follow a defensible methodology. However it is in the interaction between physical and conceptual models that the greatest scope for model improvement lies for medium to large catchments.

A near optimal disaggregation of the system response into individual conceptual components gives rise to the hope these components can be related to the important physical sub-systems and dominant processes of the original system, and that the well defined individual subsystems can be modelled using restricted physical models. Although, due to complexity (e.g. heterogeneity), it is probably not reasonable to expect accurate predictions of all parameter estimates in this way; it may be possible to obtain bounds and to predict the functional dependence of parameters on catchment and environmental characteristics. For example, recently a unit hydrograph approach was found to be inadequate for predicting the solute response of a complex catchment, even though the hydrograph response was well represented (Barnes et al., 1997). In effect, this generalisation modelled the residence time distribution of solute. A similar generalisation may be used, in principle, to model sediment residence times. Residence time model parameters, e.g. the above threshold parameter, could then be related to a physical model of hillslope storage, parameterised in terms of rainfall intensity, hydraulic conductivity, slope length and slope.

This approach to modelling catchment response preserves the apparent simplicity of observed response data, while allowing the origin and dependencies of the parameters of the model to be investigated in detail as required. The decomposition of the system is natural, in the sense that it follows from the system response itself, rather than from some preconceived notion of system behaviour. The task of characterising individual subsystems rather than the whole system in process terms is much better constrained, and processes which have no observable impact on catchment response can be ignored; while such a model can be extended in a controlled manner should further information subsequently become available.

4. Model frameworks

The conclusion from the above analysis of the three classes of models is that not only is no satisfactory general catchment model of P production available, no such model is in fact possible. In order to give a satisfactory description of P response which is transferable, and able to present the full range of management options, the model must be process oriented. But any such general physically-based model is necessarily highly parameterised, with data requirements far exceeding what is normally available, even in a research environment. Any model of this type must also be far from simple, requiring expert knowledge both for parameter assignment and operation.

Faced with these difficulties, a novel approach has begun to emerge where the search for a single universal model has been abandoned, to be replaced by the quest for an adaptable framework within which models derived for a particular set of circumstances can be built and related in a consistent way. Pioneering examples of this approach are the "models" of AEAM (Walters, 1986), and the model management approach adopted by the SWAMP project (Abel et al., 1993) for integrating water quality models. A similar approach has been taken by the ASPRU group in Toowoomba in developing a framework for testing pasture and crop models; and the TERRA group in the USA (supported by USGS, USDA-ARC and SCS) are attempting to develop model interchange standards.

Additionally, a number of the "models" mentioned previously actually represent a suite of related models, with a core framework being adapted for different purposes. For instance, HSPF has deliberately been developed in a modular fashion, with the prospect of combining modules for sediment production and transport with those for stream routing, and recent developments of the CMSS model are explicitly designed in sympathy with this point of view, with provision made for incorporating user-defined models using basic mathematical tools provided within the model itself. Similarly, TOPOG actually consists of a number of versions, with adaptions of the original version for dynamic catchment behaviour, and solute transport, for instance. These model "families" are generally based around at most a small number of major processes or features (e.g., a explicit topographic representation of the landscape for TOPOG), and are confined to a particular scale or restricted range of scales; consequently, it is very difficult to relate them to other models, written from a different viewpoint. A much more general approach is needed if the broad spectrum of available models is to be drawn into a single relationship.

A More General Approach to Catchment Modelling

Although it may be possible to incorporate arbitrary models into a general framework by adapting them to a common input and output format, classifying them according to model assumptions, this is of limited help in the main task of deciding which model to use for a particular application. A subjective decision is still required to determine which processes and what amount of detail (and hence data requirements) are necessary for the given purpose. This section outlines one possible aproach (or organisational principle) which can give guidance for model selection and development for a large class of modelling applications.

From the point of view of dynamic catchment models of P, there are three essential concepts to be considered. The system being considered, P production, and routing. Firstly, it is necesary to decide what system is being considered; this in turn determines what is to be considered as a source or sink, rather than merely an internal storage with it's corresponding effect on residence times. Secondly, the production rate of P within the catchment system has to be determined. For a point source, such as a sewage outlet, this may be given as a function of time; while for a diffuse source such as particulate-P produced by (say) sheet erosion, the erosion process needs to be characterised, possibly in terms of land use, topography, and climatic conditions. Thirdly, given the production rate, the flux of P at any point depends on the time it takes (i.e. it's velocity) between the point of origin and the point of delivery. Any dynamic catchment model of P movement will implicitly include both these components, although there may be coupling between them (e.g. the rate of particle detachment depends on the concentration of suspended particles, etc.). In any case, at a particular time, the P output of a catchment represents production at a range of different earlier times, depending on the time taken to travel from the point of production. Even for P molecules having effectively identical times and places of origin, the travel times, or residence times, will have a distribution due to different travel pathways; this distribution is called the "residence time distribution" for P. The flux of P can then be equated with the production rate at previous times, weighted according to the residence time distribution, and summed over all residence times and points of production. For viewpoints from which the production rate of P can be assumed to be spatially uniform across the catchment, it is possible to sum the residence time distribution at each point to give a catchment wide residence time distribution. Mean fluxes can be determined by summing up the production rates over the catchment area, provided the averaging period is long compared to the mean residence time (say) for the catchment; otherwise, a dynamic approach must be adopted.

What are the advantages of this sort of approach? To begin with, it allows disparate models to be related, both by focussing attention on what is assumed to be the main sources of P, and through the form of the implicit residence time distribution predicted by the model. The catchment system can itself be regarded as having the effect of a (generally non-linear) filter, causing a reduction in the variability of the input signal of P. In quantitative terms, the catchment yield of P is given as the convolution of the P input (as a function of time), with the residence time distribution which acts as a filter. In this way, the effect of source variability on yield is largely decoupled from catchment-specific effects represented in the residence time distribution. Conversely, the residence time distribution effectively integrates all the catchment processes and their interactions, to give a system wide response.

Secondly, by emphasising the importance of time in P transport, the approach gives a rationale for deciding the relative importance of different processes at different scales. The most significant processes for determining catchment fluxes of P are those which contribute most significantly to the mean residence time. Hence, for the conceptual model implicit in section 2.1 with different processes adding or subtracting from a series of internal storages, the relative importance of each term can be assessed in these terms, and for each pathway only the terms which contribute most to the residence time need be retained, leading to a simplification of the model. This provides a valuable guide for the selection of which models are relevant to a particular application. For instance, for small, relatively steep headwater catchments, the major processes affecting P transport are hillslope processes such as sheet, rill and gully erosion; and the residence time for soil particles on a hillslope is largely topographically controlled, with transport in the stream network being relatively rapid. Hence a topographically explicit model is appropriate for this application. Conversely, for a much larger catchment, with lower average slopes, the time spent in the stream network dominates, and a much simpler representation of topography combined with a more sophisticated channel routing representation is appropriate. Because the residence time distribution implicitly covers an infinite range of residence times, there is no restriction to a particular range of scales. In reality, as catchment size increases, the residence times will also increase in a "smooth" way, even though the dominant transport processes may change. In practice, it is seldom necessary to obtain the residence time distribution explicitly from a particular model, though it may well be instructive. Through facilitating the selection of the significant processes appropriate to a particular system, the residence time approach can provide a semi-objective means of discriminating between different model representations in a general model framework.

The most important part of any modelling exercise is the insight into the functioning of the system as a whole. It is this insight which allows the modeller to select between alternative models. The insight can come from accumulated experience, either first hand or from the work of others; but it is also necessary to obtain real information on the particular system under consideration, though this may take many forms (e.g., information on climate, soils, relief, etc.). Ultimately, as argued above, without actual data on the whole-system response, the modeller must necessarily make very subjective judgements about what is believed to be the dominant processes, with no reality check on these assumptions. The only way to gain real insight into the P response of catchments is through high quality data sets from a range of different catchments, with resolution sufficient to discriminate between different possibilities. At present, modellers implicitly choose a particular model representation on the basis of some supposed "similarity" between the system under consideration, and other systems previously studied, to which the model has supposedly been applied successfully. The decision about what is similar is seldom made explicit, nor is there justification for the implicit assumption that the observed similarities are in fact normative for system behaviour. The approach suggested above gives objective guidance for this selection process.

Finally, an important advantage of this approach is the questions which it forces the modeller to ask. What are the relative strengths of the major sources of P; what are the main processes affecting the response of this particular catchment, and how do they combine; how does the residence time distribution change under different conditions? The residence time framework inevitably involves dynamics of water along with phosphorus. The modeller is then confronted with the more fundamental question of the role of water flow relative to that of sediment transport, in determining system behaviour, e. g. algal blooms. These questions themselves suggest how models may be improved, what extra information would be most useful, and hence what is the most effective experimental design for obtaining further insight into system behaviour.

Summary

Whatever the type of model employed, the most important aspect from a modelling point of view is the insight into the system behaviour which is to be represented by the model. In the case of empirical models, or conceptual models involving systems identification, the insight can come from direct examination of response data. This seems preferable to the common practice of identifying relevant processes for a physical model in advance, presumably from insights obtained from previous experience with other systems identified as "similar"; the dangers of prejudice and difficulties for inexperienced modellers are obvious. From this point of view, the value of an experienced practitioner rather than a part time manager/modeller is evident, in the same way as experienced practitioners are valued in other professions and other aspects of hydrology. The only way to gain real insight into system behaviour, and confidence in model selection, is to obtain high quality, high frequency data sets at the appropriate scale, for the range of system behaviours observed in Australian catchments. Lack of such data sets is the main impediment to advances in this area as a whole, of which quantitative modelling is but a part.

It is clear from the above analysis that different models, and different classes of models, suit different needs and different systems. Where relationships are simple, empirical models can give highly accurate predictions of system behaviour, possibly in conjunction with simple conceptual models. At small scales, where process models can be characterised with some degree of confidence, and the application justifies the expense of data collection, physically based models may be warranted. In all cases some degree of simplification, involving a decision to accept less than perfect agreement with actual behaviour, is mandatory.

Recognising that no universal model is possible, an experienced modeller must have access to a range of models to suit different user needs and data avilability (or the ability to construct a purpose-built model). Ideally, relationships between models should be accessible through making assumptions explicit, with similar input and output formats to facilitate model combinations and comparisons between alternative approaches. What this requires is the further development of the concept of a model framework, in which individual models can be embedded.

In this way, different user needs can be satisfied by identifying the most appropriate type of model out of the range available. By requiring a consistent approach to data input and output, different models can be effectively related, changing needs can be efficiently met, and insights and knowledge can be accumulated in a form of knowledge system designed for expert modellers.

In this context is must be said that the widely used CMSS is fundamentally a decision support system (DSS), rather than a model, or modelling framework. A DSS gives a manager ready access to input data and outputs of the embedded model. In principle, a DSS is a tool that assists professional managers and modellers to collaborate effectively in exploring the feasibility of alternative management options. Olley et al. (1995) discuss the practicability of integrating alternative model structures, including residence time models, with CMSS. It should be even more practical to give explicit treatment to stream bank erosion as a P source, and to floodplains as depositional structures. However, such features are implemented basically by editing input data at present, rather than by specifying these concepts. Currrent DSS systems generally give the user no opportunity to investigate changes to basic assumptions directly, e.g. by interactively choosing alternative models (or their embedded options) from a model framework.

CONCLUSIONS

Toxic blue-green algal blooms have been a water quality problem in Australia for many years. However, it required world-wide publicity, caused by the occurrence of a blue-green algal bloom along ~1000km of the Darling-Barwon River in late 1991, to produce an urgency that set Federal and State Governments seeking for immediate solutions. While research on algal blooms had been carried out in Australia for many years there was little understanding of how to combat the problem on a catchment scale.

As a result Australian water managers were forced to use the limited Australian data and extrapolate from the more extensive overseas research and management techniques. Harris (1995) however, has indicated the dangers of uncritically using overseas research and management techniques for Australian ecosystems. Wasson et al. (in press), although looking at different parts of the Australian landscape to Harris, reflected the same caution with the title to their paper; "Imports can be dangerous - appropriate approaches to Australian rivers and catchments".

As well, there is the problem of yielding to quickly to the pressure (public and political) to carry out remedial action. Too often hasty remedial action to solve pollution problems has lead to a focus on the obvious sources of pollution (point sources) because they are easily identified, measured, and can be controlled by policy and regulation. However, this hasty action can lead to disappointing results over time, to public distrust, frustration with scientific research and with the methods of restoration. Non-point sources can often be the major contributor of pollution to waterways and measures aimed at point sources may have minimal effect on the overall contaminant loading, particularly in systems dominated by sediment-water interactions. While this discussion is about developing properly focused P remedial actions in Australia, similar argument has been shown to apply to other pollutants in other countries. For example, Diamond et al. (1996) found similar arguement applied for arsenic pollution in the Canadian Bay of Quinte, an 'area of concern' in the Great Lakes. Their study that found that point sources (industrial/sewage) contributed only 0.2% to the total annual load entering this bay has obvious management implications.

The data discussed in this interpretative review indicates that it is important that the unique processes/mechanisms leading to bloom formation in Australian waterbodies are understood and that at least qualitative mass balances of P loads to waterways be derived. These data are necessary to properly design long-term monitoring programs, and to indicate the short and long-term management strategies necessary to improve water quality.

Australia is an old arid low relief continent, the degree of weathering of transported sediments is high, and P is predominantly associated with the clay minerals and their iron oxyhydroxide coatings in soils (Norrish and Rosser (1983). Essentially all P in transported soils is bioavailable in waterways under the appropriate physical and chemical conditions. The hydrology of Australia shows the most temporal variability in the world (Finlayson and McMahon, 1988), characterised by extreme events of short duration (days) and long periods of little or no flow in our rivers. Storm loads transport the major P loads to our rivers, and in Australian rivers where clay-rich sediments are common (a large part of Australia) there is a strong particle-P association with generally very low concentrations of soluble P (Oliver, 1993). The climate of Australia promotes stratification even in relatively shallow waterbodies and the long water residence times lead to resolubilisation of P from bottom sediments and the eutrophication of standing waterbodies.

To survive in such an arid landscape Australians have, since European Settlement, constructed many storages and weirs along river systems, and extracted water for irrigation. Australia stores more water per capita than any other nation (DEST, in press), to secure a water supply in the world's most variable rainfall regime. This has increased water residence times and the potential for P release from bottom sediments to be a major source of bioavailable P for excess algal growth (Harris, 1995). Eutrophication is now recognised as a major problem in Australian waterways.

There are now good reasons to suggest that there are major differences between Australian fluvial systems and those in other parts of the world (Wasson et al., in press; Wasson and Sidorchuk, in press). The study of the Murrumbidgee catchment (Wallbrink et al., in press) showed that P loads in the river were dominated by natural sources. Similar results were found for the eutrophic Chaffey Reservoir (Caitcheon et al., 1995), and recent work on the Darling-Barwon River also suggests the P loads are dominated by natural sources (Wallbrink and Murray, pers. comm.). Radionuclide evidence and observation of the erosion process common in Australia (i.e. gully dominated) suggests that subsoil rather than topsoil sources dominate the sediment (and P) loads to our major river systems. Given that the major P loads to our river systems come from diffuse sources, and the fact that in clay-rich environments surface adsorption reactions are rapid, also suggests that under these conditions the distribution between the different forms of P (dissolved or exchangeable) is independent of origin. That is, the major P loads control the concentration of soluble P in the water column.

Phosphorus loads to stream systems from STP's, in in-land clay-rich environments (e.g. MDB), are small compared to diffuse loads and it is suggested that removing STP's to try to achieve lower soluble P concentrations will not be successful. As well, maintaining reasonable flow rates in Australian inland waterways is important for the ecology of stream systems, and could also be important in controlling excess blue-green algal growth. Sherman, et al. (1994) were able to calculate the minimum flow necessary to eliminate stratification on a diurnal basis and thereby light-limit the growth of cyanobacteria. Removal of STP's from many in-land Australian stream systems would be an important loss of flow.

Because the major loads of P to rivers move during storm events the ability to reduce the delivery of P to say a storage is both limited and would take decades to centuries to flush out the already deposited sediment-bound P. The inevitable conclusion from this is that for management to have a rapid effect on water quality it must be directed at the waterbody itself. This is not to downplay the importance of catchment remedial work, which can be carried out for a variety of reasons which benefit the ecosystem, and are essential for long-term reduction of nutrient loads to waterways. But for short-term improvement there is no alternative to direct management of the waterbody (e.g. destratification, environmental flows, chemical treatment, etc.).

Clearly land use affects P exports to rivers, however, in Australia there are many examples showing very large annual P exports for a small fraction of the annual flow. As well, as catchment size increases, the increase in sources and sinks in the catchment increases the complexity of sediment and P movement to stream systems. In Australia P exports can be very variable in space and time. This review has shown that the Australian data base for P exports to stream systems is very limited. A recent discussion of the use of an overseas developed model (AQUALM) to predict P exports from an Australian catchment shows that it poorly matches actual measured loads (Wasson et al., in press). New approaches to modelling P exports in Australian catchments are needed.

The challenge for the modeller is to develop a (generally quantitative) description of the system under examination which both represents the actual behaviour of the system and suggests an explanation in terms of known processes; uses observable data (appropriate to the scale of the system), and is capable of discriminating between alternative management strategies at an appropriate scale.

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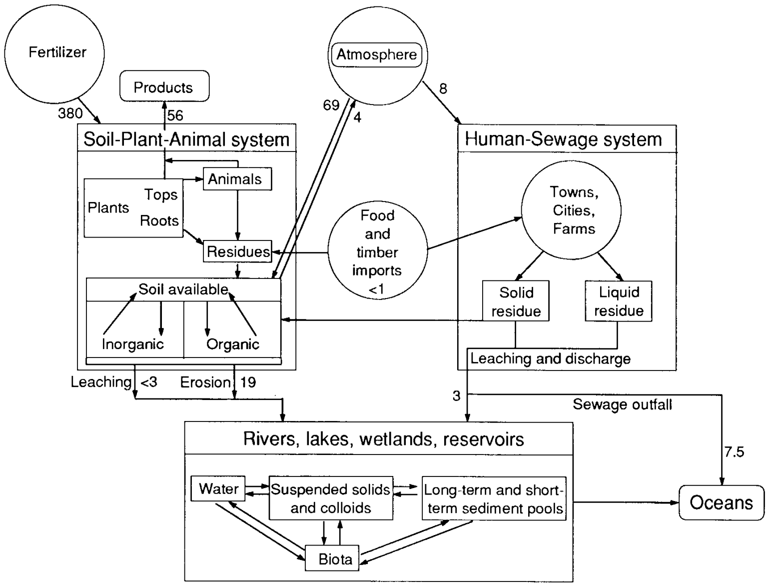

Fig. 1. The current phosphorus cycle in Australia. The circles are sources, the boxes are pools, and the polygons are sinks for phosphorus. The numbers are mean annual fluxes (kt/yr) of phosphorus.

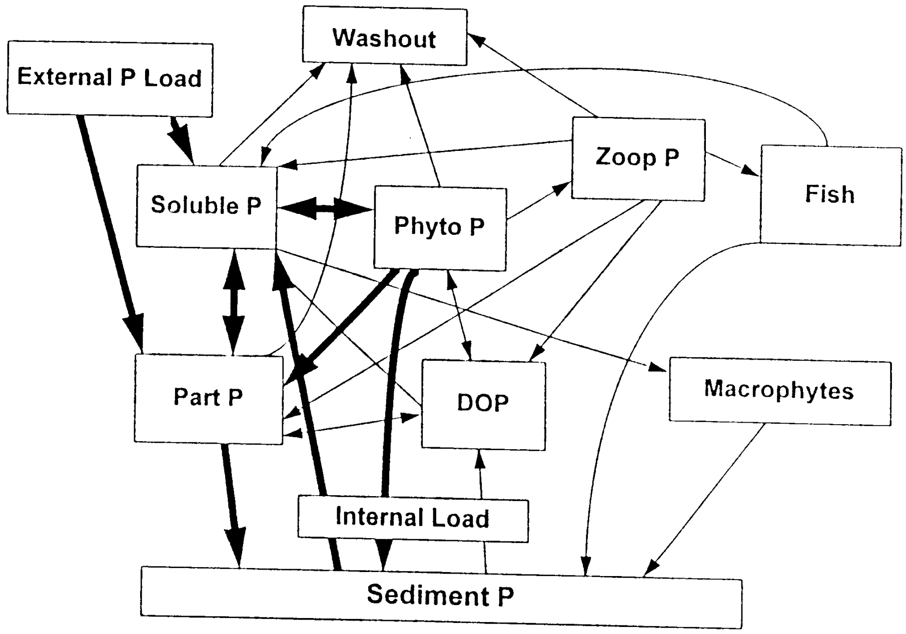

Fig. 2. Diagrammatic representation of P fluxes in a eutrophic aquatic ecosystem (Harris, 1995). The thicker connecting arrows indicate the major fluxes.

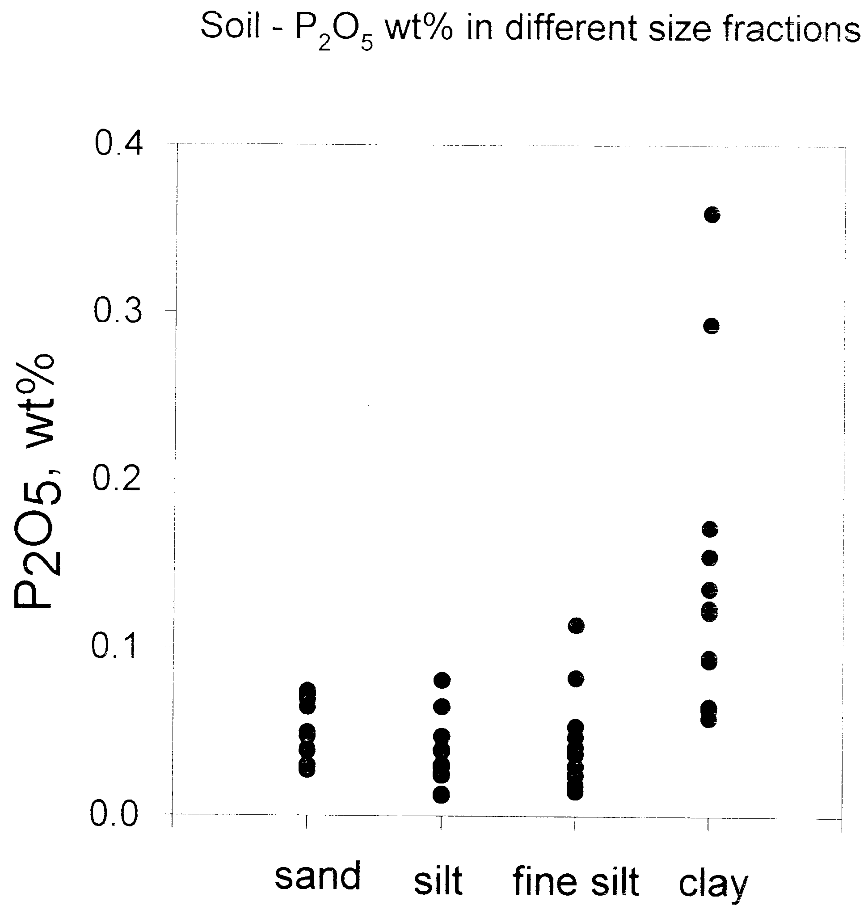

Fig. 3. An example showing that the bulk of the P (and Fe) is concentrated in the very fine clay fraction of non-aggragated soils.

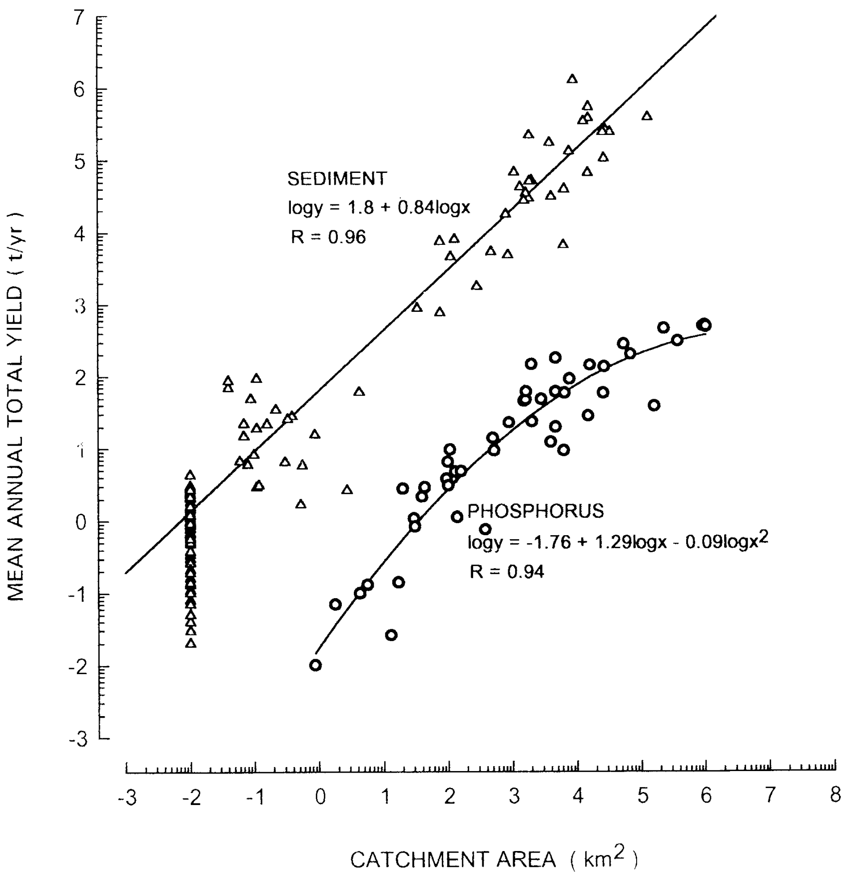

Fig. 4. The slope of the regression equation relating mean annual fluxes (of sediment and P) to a wide range of catchment areas showing that it is usually < 1.0.

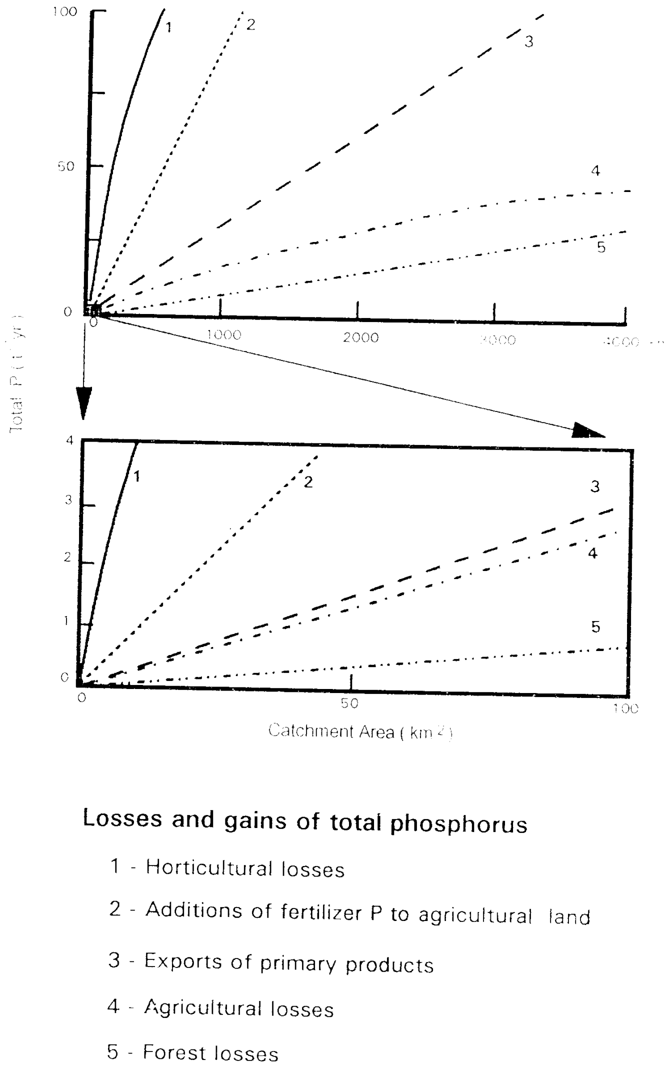

Fig. 5. Losses and gains of total phosphorus (tonnes per year), for different land uses and catchment areas.

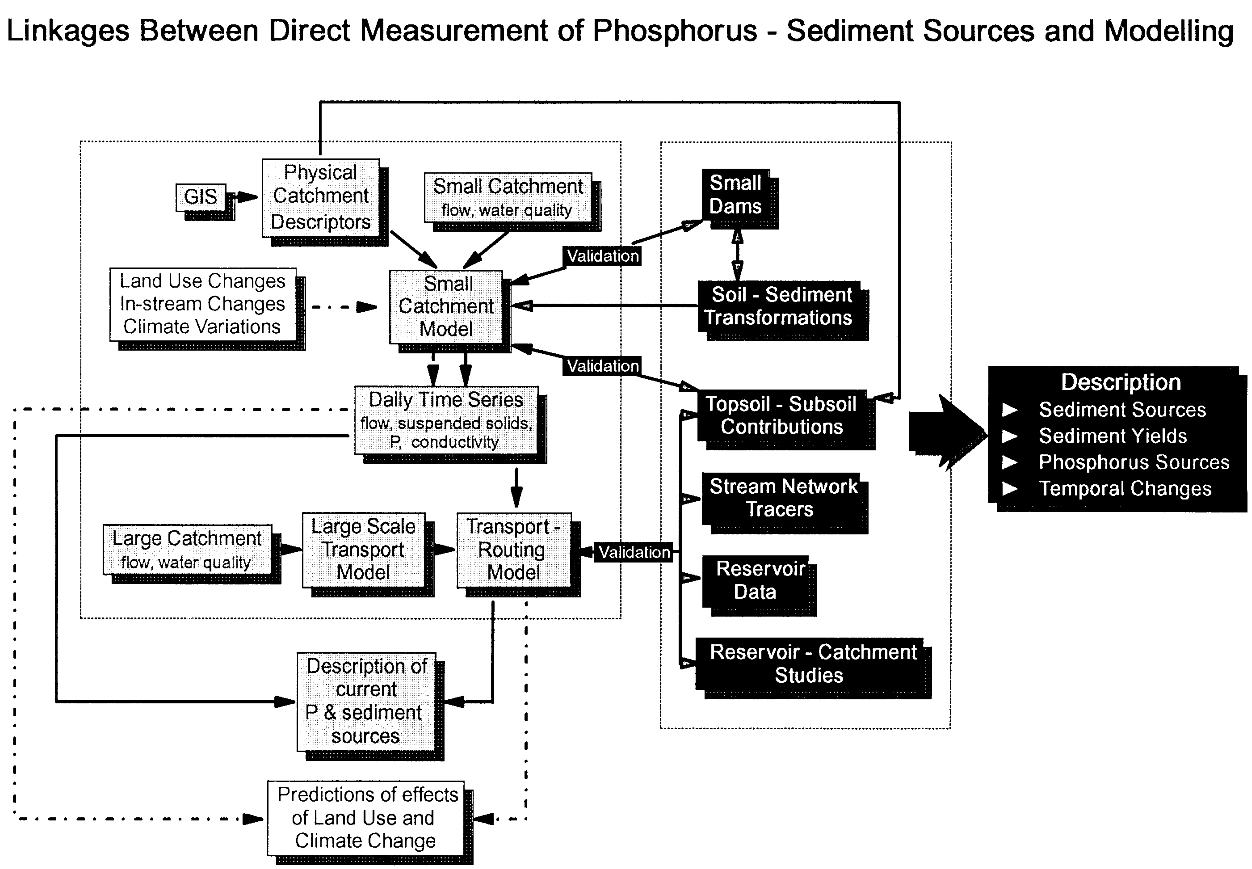

Fig. 6. Linkages between direct measurement of phosphorus-sediment sources and modelling.