32 Strategies for the Sustainable Management of Phosphorus

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In this chapter, we define sustainable agriculture as "farming systems that are environmentally sound, profitable, productive, and maintain the social fabric of the rural community," after Harwood (1990) and National Research Council (1989). A more detailed definition is given in the U.S. Food, Agriculture, Conservation, and Trade Act (1990), as "an integrated system of plant and animal production practices having a site-specific application that will, over the long term; satisfy human food and fiber needs; enhance environmental quality and the natural resource base upon which the agricultural economy depends; make the most efficient use of non-renewable resources and on-farm resources and integrate, where appropriate, natural, biological cycles and controls, sustain the economic viability of farm operations; and enhance the quality of life for farmers and society as a whole."

One aspect of sustainable farming that is becoming increasingly important is the use and management of phosphorus (P). This nutrient has played an important role in raising agricultural productivity worldwide, but adverse environmental impacts associated with its past and current use on farm land are now becoming apparent in the developed countries as production methods have intensified and farming systems have become more specialized. These concerns relate primarily to the deterioration in water quality caused by accelerated P transfer in land runoff encompassed by the term *eutrophication* (Mainstone and Parr, 2002; Foy, 2004). Recent attempts to restore semi-natural grassland ecosystems as part of European Union (E.U.) conservation policies may also be confounded by high soil P

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fertility (Critchley et al., 2002; Gough and Marrs, 1990). Large amounts of manufactured derivatives of rock phosphate are still imported into agricultural systems as feeds and fertilizers. These imports will become increasingly expensive as high-quality, accessible phosphate rock deposits become gradually depleted. More sustainable and renewable sources of P to meet the requirements of crops and livestock are required, for example the recycling of precipitated phosphates from wastewaters (Gaterell et al., 2000; Johnston and Richards, 2003).

There is, therefore, an urgent need to re-consider how to manage P inputs efficiently for a profitable agricultural industry, yet maintain a diverse range of aquatic and terrestrial ecosystems, and with due regard to the efficient exploitation of natural P reserves. This requires an understanding of the nutrient conditions that cause ecosystem dysfunction and the complex processes governing P cycling and mobility in the landscape, the interdependent linkages between farm, watershed, and ecosystem scales and the factors controlling them. These aspects are covered in detail in previous chapters, and are increasingly being evaluated as part of 'life-cycle analysis' for agricultural commodities and the production systems from which they are derived (Cowell and Clift, 1997). It is noteworthy that water quality concerns associated with nonpoint P sources from agriculture have arisen at the watershed scale rather than at the farm scale, reflecting the *cumulative* impacts of individual nutrient and land management activities on farms. Also, in many areas, the agricultural contribution to eutrophication has been, or is, masked by non-agricultural diffuse and point source P inputs to varying degrees, and involves nutrients other than P. This raises issues of scale, difficulties attributing cause and effect, and the need to consider interactions with other pollutants, which must all be taken into account when developing sustainable rural solutions to accelerated P loss from agricultural land.

The accumulation of P within terrestrial and aquatic environments is such that even if P was no longer added to agricultural systems, there would be a considerable time-lag (years or decades) before improvements in water quality, or regeneration of diverse habitats, might become apparent. In the United Kingdom (UK), the national P surplus has decreased in recent years due to a stabilization of P inputs, but there is still a trend for P accumulation (Fig. 32–1). The emphasis must therefore be on preventing further deterioration and taking strategic and sustainable actions sooner rather than later, otherwise we are simply and literally storing up more severe problems for future generations to confront. This requires a fundamental shift from current general guidance on *Good Agricultural Practice* (e.g., Department for Environment, Food and Rural Affairs, 2002) to more proactive implementation of cost-effective and targeted Best Management Practices (BMPs) (Department for Environment, Food and Rural Affairs, 2003), with mutual farmer-regulator agreement of local solutions to local problems. In turn, this will require a provision for additional farmer awareness, training, and advisory support, involve a commitment to better record keeping and farm planning, and incur variable levels of cost including capital grant support (Withers et al., 2003a).

It is recognized that a combination of BMPs, involving not just better management of P inputs but also better management of land, are required, and that these must be implemented at a sufficient intensity across the watershed to achieve desired goals (Sharpley and Rekolainen, 1997; Withers and Jarvis, 1998; Sims and

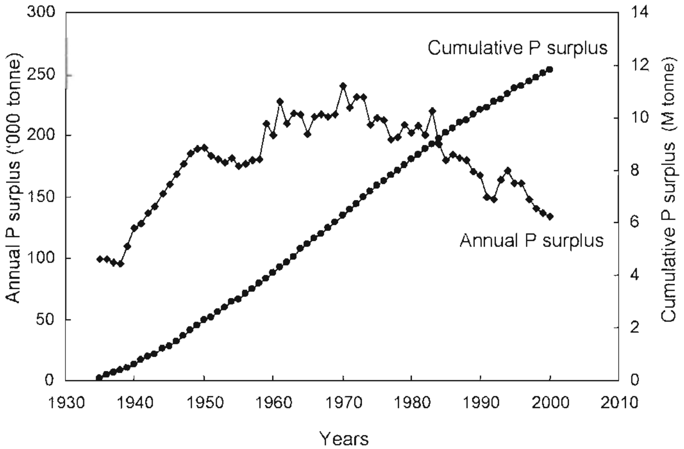

Fig. 32–1. Trends in the annual and cumulative surplus of phosphorus (P) in the UK between 1935 and 2000 (adapted from Withers et al., 2002).

Kleinman, 2004). However, it is, as yet, unclear to what extent P loss from existing land uses can be overcome by more sensitive management rather then the alternative of widespread land use change or restrictive farming. Under European Union (E.U.) proposed reforms of the Common Agricultural Policy (CAP), the role of agriculture in the rural economy is being re-evaluated. Significant areas of agricultural land may come out of production in an attempt to strike a more sustainable balance between a viable agriculture, a diverse range of habitats and good water quality. This chapter discusses how agricultural P use may become more sustainable within the confines of production and environmental pressures, the potential measures and mechanisms by which this might be achieved, and how science can be rapidly and effectively incorporated into agricultural management policy.

AGRICULTURAL PRODUCTION TRENDS AFFECTING SUSTAINABILITY

Significant changes in agricultural production have occurred over the last 75 yr which have influenced the usage, cycling, and accumulation of P in the landscape, and increased the risk of accelerated P transfer in land runoff to inland and coastal waters. Prior to World War II, farming communities tended to be self-sufficient. Home-grown feeds were recycled to meet animal requirements ensuring a sustainable food chain (left part of Fig. 32–2). With the advent of new technologies, mechanization, increased chemical use, specialization, and government incentives, agricultural production has more than doubled and become concentrated on less agricultural land and on fewer, but larger, farms (Evans et al., 1996; Withers et al., 2002). For example, between 1950 and 1990, U.S. farm land has decreased from 1200 to 970 million acres (20%) and the number of farms from 5.6

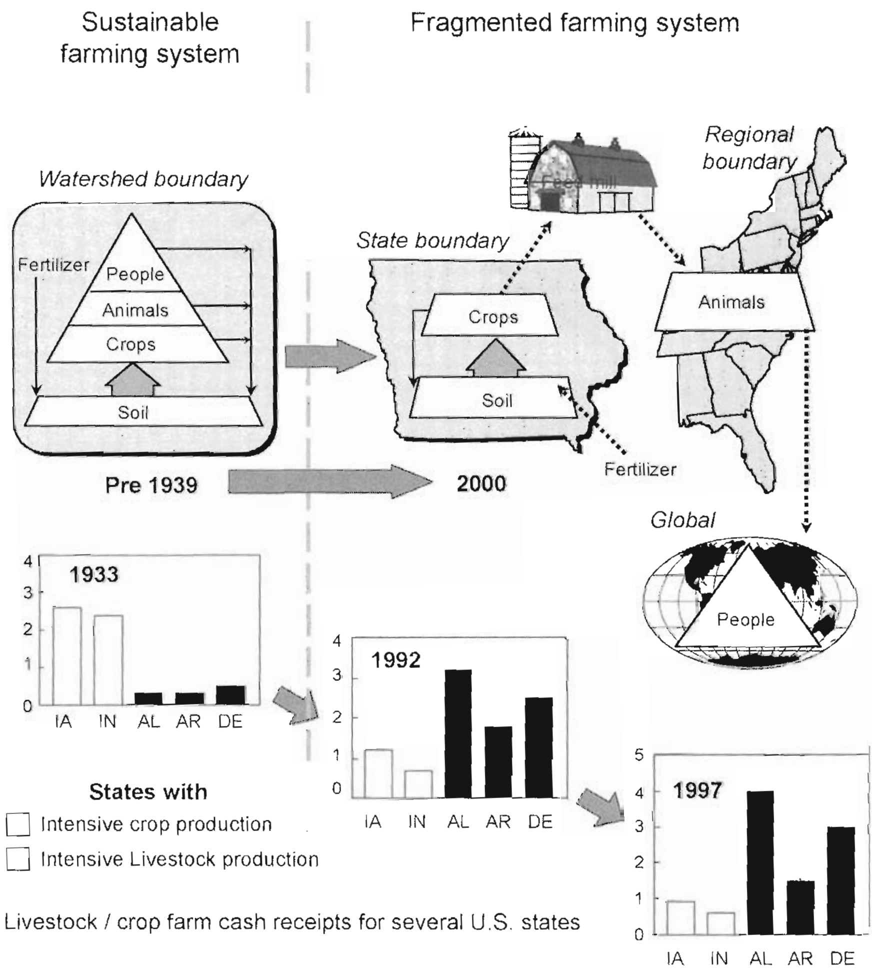

Fig. 32–2. Relationship between changes in the spatial distribution of farming practices and the ratio of livestock/crop receipts for 1933, 1992, and 1997 (adapted from Lanyon, 2000).

to 2.1 million acres (63%), while average farm size has increased from 213 to 469 acres (120%).

In many U.S. states, Animal Feeding Operations (AFOs) are now the major source of agricultural income. Farmers have turned to animal confinement (i.e., as AFOs) because of industrialization processes affecting the agricultural economy, including new technologies for production and marketing, competitive pressures, and the need to respond to consumer demand for quality meat or milk products at a low cost. These processes are driven by firms seeking profits available through adoption of new technologies for farm production or marketing and specialization (Abdalla, 2002). At the farm level, these changing factors are telling farmers to either "get big or get out" (Abdalla et al., 1997). The industrialization of U.S. agriculture is bringing about much change. In animal agriculture, the concentration of animals on fewer, larger farms and new marketing arrangements are changing the scale of farming and leading to the growth of AFOs.

In addition, farm income from traditional grain crops has decreased. For example, income from livestock operations in Alabama ($2.5 billion) and Delaware ($0.52 billion) were a respective four- and threefold greater than from cropping ($0.6 billion for Alabama and $0.17 billion for Delaware) at the time of the 1997 U.S. census (Fig. 32–2). The rapid growth of the animal industry in certain areas of the USA has been coupled with an intensification of operations. For example, current census information shows that there has been an 18% increase in pig (*Sus scrofa*) numbers in the USA over the last 10 yr along with a 72% decrease in numbers of farms. Over the same 10 years, the number of dairies has decreased by 40%, but herd size has increased by 50%. A similar intensification of the poultry (*Gallus domesticus*) and beef (*Bos* spp.) industries has also occurred, with 97% of poultry production in the USA coming from operations with more than 100 000 birds and over a third of beef production from <2% of the feedlots (Gardner, 1998). As a result of this regional intensification, modern farming systems have become fragmented with increasing separation of crop and livestock production, even across regional boundaries (right part of Fig. 32–2), with consequences for P transfer. Similar trends in historical changes have been noted in Europe (Breimyer, 1962; Withers et al., 2002).

Averaged for several developed countries, only 34% of fertilizer and feed P inputs is removed in crop and animal produce, resulting in an annual P surplus of 26 kg ha^{-1} averaged over the total utilizable agricultural land area (Table 32–1). This low overall recovery of P occurs despite the relatively efficient recovery of P in crop production of up to 80%. Thus, the efficiency of P use in agriculture is dominated by animal production, as most of the crops produced (70–95%) are used for animal feed and, together with imported feeds, are poorly utilized by animals (<30% of P fed; Poulson, 2000; Valk et al., 2000). Consequently, agricul-

Table 32–1. Phosphorus (P) balance and efficiency of plant and animal uptake of P for several European countries and the USA in the early 1990s (data adapted from Iserman, 1990 and National Research Council, 1993).

Country	Area in agriculture	Input		Output		Surplus
		Fertilizer	Feed	Animal	Plant	
	10^6 ha	kg P ha^{-1} yr^{-1}				
Belgium	13.7	25	46	10	20	41
Denmark	2.8	16	49	7	22	36
E. Germany	6.2	25	6	3	1	27
W. Germany	12.0	27	10	10	3	24
Luxembourg	1.3	25	22	12	10	25
Ireland	5.7	11	1	3	1	8
Netherlands	2.3	18	44	17	5	40
Switzerland	1.1	22	11	6	4	23
United Kingdom	18.5	9	3	2	1	9
United States	394.7	39	5	13	5	26
Average	46	22	20	8	7	26

Table 32–2. Phosphorus (P) flows for four farming systems each of 30 ha with varying numbers of animals in Pennsylvania (data adapted from Lanyon and Thompson, 1996; Bacon et al., 1990).

Farming system	Number of animals	Input: Fertilizer	Input: Feed	Output	Surplus
		kg P ha^{-1} yr^{-1}			
Cash crop†	0	22	0	20	2
Dairy (Holsteins)‡	65	11	30	15	26
Hog¶	1 280	0	104	66	38
Poultry§	75 000	0	1 560	530	1 030

† 30 ha cash crop farm growing corn and alfalfa.
‡ 40 ha farm with 65 dairy Holsteins averaging 6600 kg milk cow^{-1} yr^{-1}, 5 dry cows, and 35 heifers. Crops were corn for silage and grain, alfalfa and rye for forage.
¶ 30 ha farm with 1280 hogs; output includes 45 kg P N ha^{-1} yr^{-1} manure exported from the farm.
§ 12 ha farm with 74 000 poultry layers; output includes 8 kg P ha^{-1} yr^{-1} manure exported from the farm.

tural systems that include AFOs determine the magnitude of P surpluses on farms depending on the type and intensity of livestock operations and the land area available for spreading of the manure produced (Table 32–2). As the intensity of animal production increases, more P must be recycled, the P farm surplus (input-output) becomes greater, soil P levels increase and the overall risk of P loss in particulate and dissolved forms increases (Fig. 32–3 and Haygarth et al., 1998; Sharpley, 2000; Withers et al., 2002). More site-specific data on P export associated with increased soil P and P inputs are given in earlier chapters of this monograph, and by Haygarth and Jarvis (1999), Sharpley and Rekolainen (1997), Sims et al. (1998), and Withers et al. (2003b).

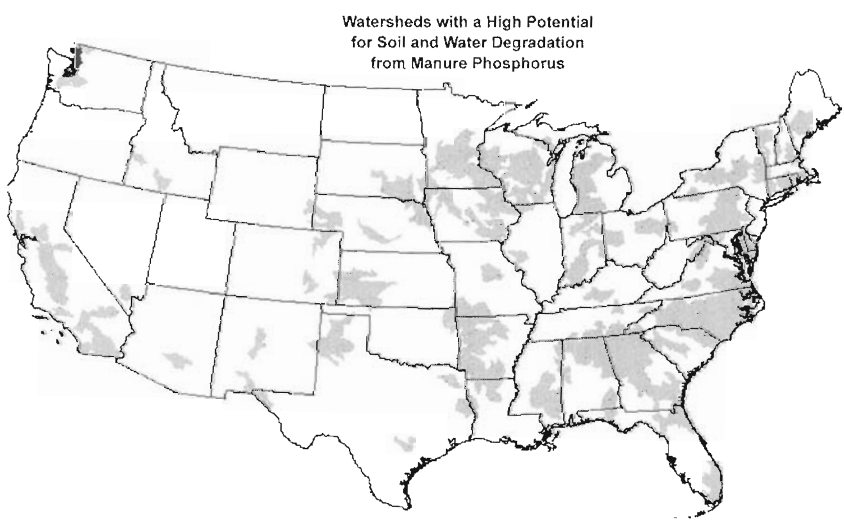

Fig. 32–3. Watersheds with a high potential for soil and water degradation from manure phosphorus (P) (adapted from Kellogg and Lander, 1999).

The location and size of livestock production systems can have a large impact on the flows of P at regional scales as well as at the farm scale. Using the Chesapeake Bay drainage basin as an example, if all the manure P available for application in 1933 were applied to corn (*Zea mays* L.), large areas of the basin would not have received adequate amounts of P to meet crop requirements (ca. 30 kg P ha^{-1}; Fig. 32–4). There were some small areas to the north where manure P was in excess, probably due to limited areas of corn production at that time, but without importation of fertilizer from outside the basin, the availability of manure was limiting total crop production over the watershed. By 1992, manure P exceeded the corn requirements (>60 kg P ha^{-1}) in large areas of Virginia, West Virginia, Delaware, and parts of Pennsylvania and the New York area of the drainage basin (Fig. 32–4). In practice, there is little export of manure off the farm, and

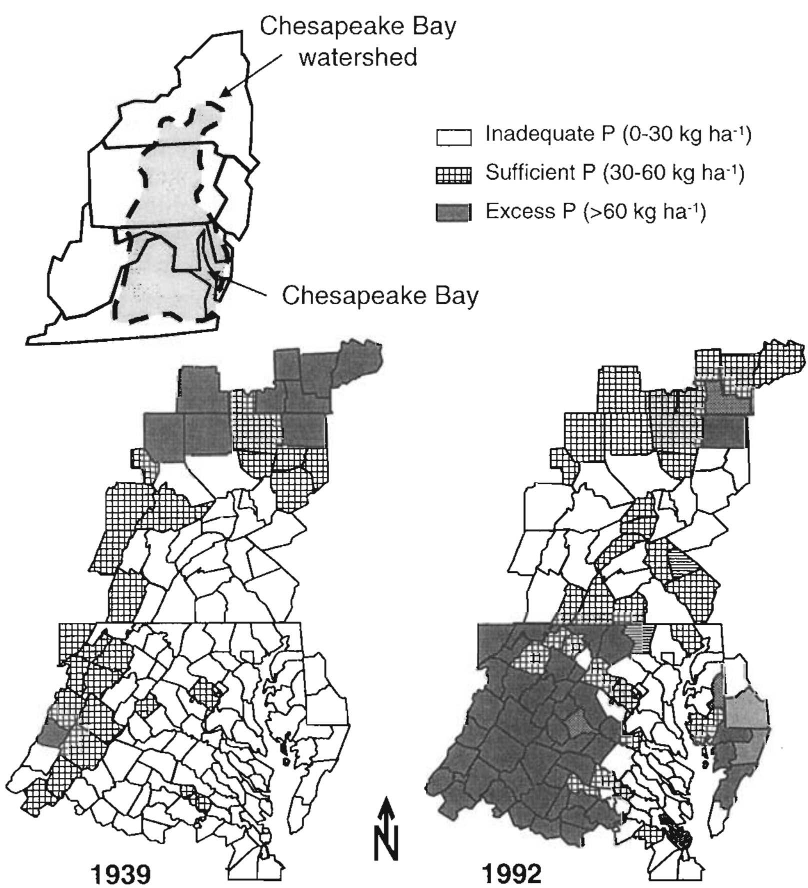

Fig. 32–4. Available manure phosphorus (P) per acre of corn in the Chesapeake Bay drainage basin in pre- and post WWII years (1939 and 1992, respectively, adapted from Lanyon, 2000).

areas with AFOs, where P is accumulating, become spatially separated from those areas which still require P inputs for profitable crop production. The increasing separation of crop and livestock production has also affected the scale and intensity of production and the geographic location of agricultural production activities (Breimyer, 1962; Lanyon 1995). In the USA for example, the upper Midwest (Iowa and Indiana on Fig. 32–2) shifted from animal to crop agriculture and the Northeast, Southeast, South Central, and Great Plains favored animal agriculture as a result of industrialization. Similar trends in agricultural intensification, specialization, and fragmentation have occurred throughout the developed world to varying extent, resulting in an uneven distribution of P surpluses, variable rates of accumulation of surplus P in the soil and increased risk of nonpoint P transfer.

It is not only agricultural systems that have intensified but also changes in agricultural methods and practices encouraged by the need to remain profitable. These have increased either the vulnerability of the landscape to runoff and erosion, the risk of mobilization of fresh P inputs or the runoff connectivity between the field and the watercourse (Sharpley and Withers, 1994; Withers and Lord, 2002). These changes include (i) removal of natural barriers to runoff such as hedgerows producing larger fields with long slopes facilitating erosion; (ii) shift towards continuous arable cultivation affecting soil organic matter contents and soil stability; (iii) preference for crops (e.g., winter cereals, maize) which leave the soil surface without crop cover for a significant proportion of the winter; (iv) introduction of tramlines which encourage and concentrate runoff along overland flow channels, which typically have very little crop cover (tramlines are dedicated wheeling areas at regular intervals [12 or 24 m] in the crop that allow chemicals to be repeatedly applied without compacting the rest of the field); (v) use of larger, more specialized machinery capable of faster work rates which encourage soil compaction, deepen cultivation depth and reduce soil structural stability; (vi) switch from solid straw-based livestock waste handling to slurry-based systems which increase manure P solubility and the volumes of manure to be recycled; and (vii) installation of subsidized field tile drainage encouraging preferential subsurface flow directly to the watercourse and bringing into production marginal land in high rainfall areas with a high P loss risk; and (viii) increased reliance on contractors who have less control over the timing of agricultural activities. For example, in the Baltic States, the fluctuation in P loss according to changes in livestock density since the 1960s was coincident with the introduction of tile drainage (Sileika et al., 2002). The specific impacts of these management changes will vary depending on the type of farming system, the specific management practices adopted and the inherent vulnerability of the landscape to P loss (Withers and Lord, 2002).

The fundamental changes in the infrastructure of agricultural systems and adoption of intensive agricultural practices occurring in recent decades are the result of a number of complex and interrelated factors and not merely due to independent farmer decisions (Lanyon, 2000). They have largely come about without any consideration of environmental impacts and have, in effect, decreased the long-term sustainability of P management because they often cannot be easily, cheaply, or quickly reversed. In many problem areas, intensification has occurred without consideration of the capability of the landscape to sustain modern farming methods.

The industrialization of U.S. agriculture is bringing about change in the organization of agriculture and size and location of farms. Farm structure is generally evolving from many dispersed integrated crop-livestock farms to fewer larger farms that specialize in production of poultry and livestock. Animal producers are more closely integrated into marketing functions and tend to be located in clusters near processing or infrastructure specialized to their needs. As the scale of operations has increased and production has become geographically concentrated, the potential burden placed on local environments by animal waste has increased (Pagano and Abdalla, 1994).

The regional adoption of intensive pig and poultry units on sandy soils with a limited P sorption capacity (e.g., Delaware, The Netherlands, and Belgium) is a clear example (Sims and Kleinman, 2004). A key factor that emerges is the dominant role of farm management in the P loss process, and whether the longer-term effects of progressive intensification can now be tempered by short-term management practices which seek to control the mobilization and delivery of accumulated soil P and current P inputs. It is also clear that the cycling of P in agriculture, the imbalance of P flows across geographical areas, and the capability of the land to support a particular farming system, need to be addressed by agricultural policy in order that P sustainability can be achieved over a range of scales; that is, farm, watershed, regional, and national. A key challenge to achieving sustainability is the definition of what is an acceptable level of P loss for a particular ecosystem and what controls need to be implemented to realize this reduction.

DEFINING REFERENCE CONDITIONS TO IDENTIFY P-RELATED IMPAIRMENT

Recently, the U.S. Environmental Protection Agency (USEPA) (1996a) and U.S. Geological Survey (1999) identified eutrophication as the most ubiquitous water quality impairment in the USA. In response to this problem, the USEPA established the National Regional Nutrient Criteria Program in the Office of Water (USEPA, 2001a). Part of the Office's mandate is to identify water impairment to prioritize and target remediation. This necessitates the definition and quantification of what actually constitutes impairment of a watercourse or body. Similarly, in Europe, the restoration, protection, and maintenance of 'good' water quality is a key goal in establishing a sustainable agriculture. The recently introduced Water Framework Directive (WFD) requires the setting of 'reference conditions' for different water body types to define how far they have become impaired (European Commission, 2000). Reference condition is defined as 'no or only very minor alterations' resulting from human activities to the water body's physiochemical, biological, and hydrological properties. The deviation from reference condition is defined by the Ecological Quality Ratio (EQR) which represents the relationship between current observed conditions and reference conditions, and should be a numerical value between 0 (bad status) and 1 (high/reference status). For P, the ratio is defined as the reference total P concentration/observed total P concentration. In practice, good ecological status will be defined by numerous chemical and biological criteria, whose interaction in terms of ecological impacts is still far from clear, but reference conditions are anticipated to be close to those operating in pristine waters. Clearly, nutrient status is only one of many factors influencing the quality of aquatic ecosystems.

In the USA, background or reference eutrophication levels of pristine stream, lakes, reservoirs, and other surface waters are defined by monitoring total P, total N, chlorophyll-*a*, and clarity, in waters draining areas with minimal human impact (Gibson et al., 2000). Because it can be argued that most, if not all, lakes have been impacted by human activity to some degree, reference conditions realistically represent the least impacted conditions or what is considered to be the most attainable conditions. The following guidelines are used to define minimally impacted lakes; watersheds with <1% urban land use, >65% of lakeshore has at least 10 m of natural bank-side vegetation, no point source discharge inputs, and there is no control of water level fluctuations (Gibson et al., 2000; Heiskary, 1989). This approach makes it possible to demonstrate that minimally impacted waters do in fact exist for that landscape type and locale, so that management efforts for each geographic (ecoregional) area can be better justified and linked to attainable goals (Omernik, 1987). Background levels of total P for freshwater systems in the continental USA are shown in Table 32–3, and similar approaches have been taken in Australia and Europe (Sparrow et al., 2000; Withers and Lord, 2002).

In Europe, all waters have become impacted to some extent under human settlement, and impairment can be difficult to define. Alternative approaches must then be adopted; these might include taking the lower 25% of current total P observations, inferring an historic value from diatom accumulation in lake bottom sediments (Anderson, 1997); P export coefficient modeling calibrated to a pre-agricultural reference year (Johnes et al., 1996) or the morphoedaphic index which

Table 32–3. Background total phosphorus (P) concentrations for each of the aggregated nutrient Ecoregions in the USA for freshwater systems (adapted from U.S. Environmental Protection Agency, 2001a).

Aggregated ecoregion		Total P (μg L^{-1})	
Number	Description	Rivers and streams	Lakes and reservoirs
I	Willamette and Central Valleys	47	—
II	Western forested mountains	10	9
III	Xeric West	22	17
IV	Great Plain grass and shrub lands	23	20
V	South Central cultivated Great Plains	67	33
VI	Corn Belt and Northern Great Plains	76	38
VII	Mostly glaciated dairy region	33	15
VIII	Nutrient poor largely glaciated Upper Midwest and Northeast	10	8
IX	Southeastern temperate forested plains and hills	37	20
X	Texas-Louisiana Coastal and Mississippi alluvial plain	128†	—
XI	Central and Eastern forested uplands	10	8
XII	Southern Coastal Plains	40	10
XIII	Southern Florida Coastal Plains	—	18
XIV	Eastern Coastal Plains	31	8

† This high value may be either a statistical anomaly or reflects a unique condition.

relates total P to lake alkalinity (Vighi and Chiaudani, 1985). All approaches have their limitations and of course cannot be easily validated (Carvalho et al., 2003; Foy, 2005, this publication).

The WFD requires an assessment of the impact of anthropogenic activity on the status of E.U. waters, in order that impairment can be traced back to a source. Carvalho et al. (2004) have undertaken an initial assessment of the risk of lakes in Great Britain failing to meet 'good' water quality status with respect to total P. The criterion for good status was set as a threshold based on a doubling of reference total P concentrations (i.e., good status is achieved with an EQR value (reference/observed) >0.5). The impact of anthropogenic activity was assessed by modeling the point and nonpoint P pressures (P loads) on the lake, and converting this to a lake total P concentration using the equations outlined in Organisation of Economic Cooperation and Development (1982), taking into account the sensitivity (e.g., depth and alkalinity) of the lake system. The predicted (modeled) in-lake total P concentration was then compared to the reference total P concentration for the lake type to calculate the EQR. Using this approach, it was predicted that more than 50% of lakes >1 ha in size in Great Britain (approximately 14 000) were 'at risk' of failing to achieve good ecological status (EQR < 0.5), but with large variation within different lake types. For example, the risk assessment predicted that >90% of shallow, medium alkalinity (largely mesotrophic) lakes would be 'at risk' of failing good status (Carvalho et al., 2004).

The difference between background reference levels of P and the current ambient P status provides an assessment of the extent of P reduction required within a watershed to achieve 'good' water quality, or return to a 'natural' state. This in turn informs the planning and targeting of watershed scale BMPs. In the USA, the background levels have regulatory applications under the Federal Clean Water Act and EPA expects states to develop nutrient standards, National Pollution Discharge Elimination System (NPDES) permit limits, and Total Maximum Daily Loads (TMDLs). The levels can also be used for voluntary planning and evaluation purposes (USEPA, 1998). Within Europe, countries have governmental powers to implement the WFD. Water quality goals will be achieved through river basin management planning. River basin plans must be in place by 2009 for implementation over the subsequent 5 yr.

A watershed can be divided into constituent subwatershed land units and the goal of a particular background reference P level parceled out among the tributary systems. Subsequently, individual farmers can target P load amounts as their equitable share of the remedial effort. This is subject to considerable spatial and temporal variability and thus, contention or debate; and includes an understanding of transport potential (overland and subsurface pathways), soil, fertilizer and manure management, BMPs already in place, and landscape topography and position within the watershed, and proximity to P-sensitive waters.

SHORT-TERM REMEDIAL OPTIONS

Sustainable management of P must encompass the need to bring into closer balance P inputs and outputs at the farm, watershed, or even regional scales, and the need to adopt more sensitive nutrient and land management practices to reduce runoff, particulate, and dissolved P loss from individual fields, and across the watershed. The risk of P loss within a watershed is not evenly distributed, but tends to be site specific depending on topography, soil type, land use, and management, and how well connected hydrologically the field is to the watercourse. Certain landscapes (e.g., high rainfall, steep slopes, tile drained land) and certain farming practices (e.g., intensive livestock rearing, cultivation leading to exposed and compacted soils) have a higher risk of P loss than others, requiring a targeted approach to BMP implementation. Similarly, some areas of the country or state, require more rapid implementation of control options than others, because of the immediacy of the problem, for example areas of deteriorating conservation status or rivers of high sensitivity to nutrients. The principle of targeting based on risk assessment therefore plays a key role in sustainable P management: targeting of priority waterbodies, targeting of priority subwatersheds, and targeting of BMPs to those areas of the watershed that are contributing most P.

The areas of a watershed to be targeted for BMPs will depend on an understanding of the nature of the P loss and the distribution of P loss risk across the watershed. In the USA, P control measures are targeted at critical source areas (CSAs) based on watershed research showing that the majority of the loss originates from only a small proportion of the watershed, the 80:20 rule. These are essentially P hotspots with active hydrological connectivity by overland flow (Pionke et al., 1996). In the Schuitenbeek watershed in The Netherlands, hotspots lead to high local P losses (up to 14 kg P ha^{-1}), but the majority of the watershed P load comes from a relatively large area of medium-risk land which exports much less; 2 to 4 kg P ha^{-1} (Schoumans and Chardon, 2003). In this watershed, subsurface leaching is the main pathway of loss because of the widespread build-up of soil P, and clearly P loss control strategies will not be targeted just at the hotspots. Watershed-based decision support systems for sustainable P management must therefore consider the spatial and temporal distribution of P loss risk at both farm and watershed scales. At the farm scale, the distribution of risk will determine farm management response, which conceptually might be tailored to the degree of risk (Table 32–4). At the watershed scale, the distribution of risk defines the intensity of agricultural activity impacting on the waterbody, and degree of land management, or land use change required to achieve water quality objectives.

In the UK, a twin-track approach to diffuse agricultural pollution control is being considered (Department for Environment, Food and Rural Affairs, 2003). The first track seeks to proactively implement specific solutions in high priority 'sensitive' areas based on detailed problem appraisal and monitored outcomes. Targeting of specific solutions to problem areas has a number of advantages; it helps secure practical action where it is most urgently needed, provides a basis for assessing the impact of control options through monitoring and helps to meet national legislative commitments. The second track seeks countrywide adoption of basic good nutrient and land management practice to prevent accelerating P loss and further deterioration in water quality in less vulnerable areas and in the longer term. It provides a mechanism for improving the general level of environmental management on farms and provides a lever for adoption of those targeted solutions that can be easily and widely adopted at minimal cost (Fig. 32–5; Department for Environment, Food and Rural Affairs, 2003).

Table 32–4. Conceptual scheme of management responses according to the level of risk of phosphorus (P) loss from fields (adapted from Withers et al., 2003a).

	Risk of P loss from individual fields			
	Low	Medium	High	Very high
Definition	No rapid hydrological connectivity to the watercourse	Indirect hydrological connectivity to the watercourse	Direct and rapid hydrological connectivity to the watercourse	Prolonged hydrological connectivity to the watercourse
Management strategy	No restrictions	Risk can be easily overcome by best management practices (BMPs)	Land use restrictions apply if risk cannot be overcome by BMPs	Non-agricultural use
Measures required	Basic good practice	Low cost measures usually adequate	Higher cost measures may be required	Non-agricultural use

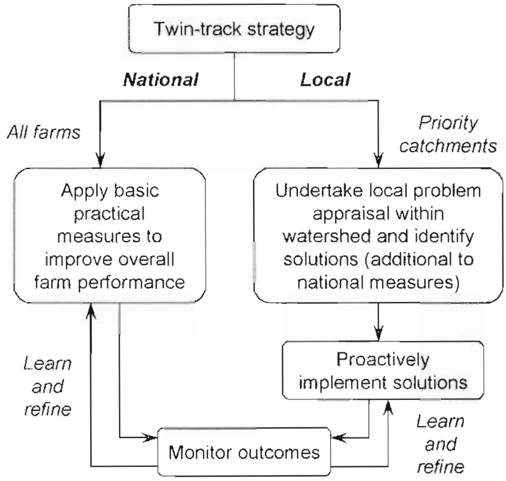

Fig. 32–5. The twin track policy strategy for diffuse pollution control (Department for Environment, Food and Rural Affairs, 2003).

The need to consider nutrient, soil, and crop management in developing sustainable use of P requires a commitment to farm, watershed, and regional planning (Fig. 32–6). This in turn requires: (i) good record keeping to audit the use of farm resources (nutrients, soil, and water); (ii) access to decision support tools to undertake nutrient budgets, choose optimum soil cultivation systems and identify problem fields; (iii) advice on the effectiveness and practicality of a range of mitigation options for controlling P loss; and (iv) a higher level of skill to cope with the increased complexity of decision-making.

Best Management Practices can be grouped into input, source, and transport measures (Sims and Kleinman, 2005; this publication). Briefly, input measures focus on customizing feed rations to meet animal requirements and amending feed

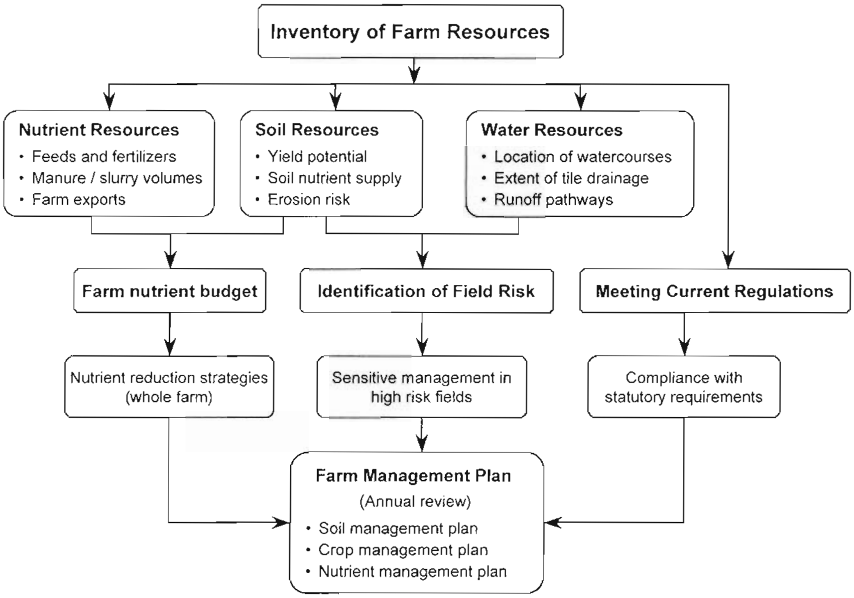

Fig. 32–6. Schematic diagram of the component parts of a farm management plan to control phosphorus (P) loss.

(enzymes) to enhance P utilization, without reducing production (Wu and Satter, 2000). Fertilizer P inputs must also be customized to match crop offtake taking full allowance of any contribution from animal manures applied and residual soil P fertility. Some fine tuning of rations may be possible based on the animals age and production value (e.g., milk, meat, breeding). More information is needed in this area. In particular, the question of whether crop fertilization should be targeted to meet the need of the crop only, or the diet of the animal must be answered. It must be recognized that reducing inputs is unlikely to have a large impact on P loss in the short term, except in highly intensive livestock systems where recycling of P under unfavorable environmental or climatic conditions would otherwise lead to high 'incidental' P losses in rapid runoff (Dwyer et al., 2002; Withers et al., 2003b). This is because there is a 'bank' of residual P already present in the soil, and as soil P levels change only slowly, losses of P in systems where soil P loss is the dominant process would only be reduced over a longer period of lower P inputs (i.e., >10 yr; McCollum, 1991; Pierson et al., 2001; Withers et al., 2000).

Source measures involve determining the correct rate, timing, and method of P application as fertilizer or manure to meet crop nutrient needs, to avoid P build-up in the soil profile and reduce the risk of 'incidental' P loss occurring directly after application. Manure testing and nutrient applications based on soil test P recommendations can be followed to ensure that P inputs are as evenly distributed across the farm as possible, thereby avoiding hotspots of P build-up. However, where on-farm P in manure exceeds crop requirements, additional measures are needed. This involves dietary modifications to minimize the amount of P to be recycled, exporting of manure to neighboring farmers with little or no P surplus, and manure (alum, Fe, waste-water treatment residuals) and soil amendments (fly ash) to reduce P solubility and potential release to overland and subsurface water flow. Additionally, there is a need to explore and develop alternative technologies and uses for manure, even if these may have only localized applicability. For example, conversion of manure to value-added products such as compost, pelletizing, and use as an energy source, such as gasification and methanol (U.S. Environmental Protection Agency, 2001b). There is also a need to determine how to facilitate or encourage markets so that alternative technologies and uses become economically viable to farmers and society.

Innovative technologies are needed to minimize P transport in overland and subsurface pathways via conservation tillage and crop residue management, buffer strips, riparian zones, terracing, contour tillage, and cover crops. For example, are there alternative plant species in buffers that can provide farm income and increase wildlife diversity, and can Fe-based compounds be targeted as buffers to reduce P in subsurface and lateral flow? Basically, these practices reduce rainfall impact on the soil surface, reduce runoff volume and velocity, and increase soil resistance to erosion. In addition, stream bank protection and fencing, buffer or riparian areas, impoundments (e.g., settling basins), and reservoirs can remove dissolved P and trap particulate P after leaving the field but before it enters P-sensitive surface waters. In some priority areas more fundamental changes in farm infrastructure may be required. These might include, for example, taking certain fields out of high risk cropping, switch from arable farming to grass or a reduction in stocking rates on the farm and will have larger financial impacts on the farm business (Dwyer et al., 2002).

Despite these advantages, any one of these measures should not be relied upon as the sole or primary means of reducing P losses in agricultural runoff (Withers and Jarvis, 1998). These measures are generally more efficient at reducing sediment P than dissolved P. Also, P stored in stream and lake sediments can provide a long-term source of P in waters even after inputs from agriculture have been reduced (McDowell et al., 2002). As a result, the effect of remedial measures in the contributing watershed may be slow, emphasizing the need for immediate action to avoid prolonging water quality problems. Overall, the key to effective remediation of P losses is through the identification of critical P inputs, sources, and pathways to prioritize and target the most cost-effective BMP for implementation, followed by a monitoring and maintenance program to ensure they continue to operate as designed (Sharpley and Rekolainen, 1997; Withers and Jarvis, 1998).

LONG-TERM POLICY MECHANISMS

Extension Education

Development of effective extension and education programs play an important role in improving farmer awareness of P loss as an environmental issue, in demonstrating and implementing sustainable P management, and overcoming barriers to uptake (Table 32–5). Some farmers are reluctant to admit that nutrient loss originates from their farm, even though they accept there are water quality prob-

Table 32–5. Potential barriers to uptake of management options for sustainable use of phosphorus (P) (adapted from Dwyer et al., 2002; Withers et al., 2003a).

Potential barriers	Description
Awareness	Lack of awareness by farmers of eutrophication problems, and potential P losses from farmland, in terms of their existence, nature, causes, solutions, financial impact, legislation, and impending regulatory controls.
Skepticism	Farmer skepticism as to effectiveness and/or legitimacy of policy mechanisms, distrust of certain information sources or to the need to manage P inputs. For example, overregulation and high administration costs.
Willingness	Lack of willingness to act by farmers because they either do not take responsibility for contributing to the eutrophication problem, or do not consider it important enough or directly relevant to the farm business. Clearly relates to lack of awareness and perception above, but also includes willingness to overcome the 'hassle' factor, and adopt a sustained attention to detail in farm management.
Ability	The limited ability of farmers to plan, manage, and implement certain management options, without specialist training, advice, or equipment. A higher level of skill is required in using farm decision support tools, understanding some recommendation systems, or using specialist equipment as part of sustainable strategies for P use.
Practicality	The practicality of various P management options to suit different site situations. This will differ according to the particular farm and requires integration of farmers own ideas. Options need to be realistic, practicality needs to be demonstrated and linked to effectiveness.
Cost	The cost of implementing different management measures across a range of farm types, particularly during periods of low profitability from agriculture generally. Some options may require access to capital grants to change farm infrastructure where the farmer has to make a financial contribution.
Effectiveness	The effectiveness of the suggested management options in controlling diffuse agricultural pollution. Linked to lack of research data and demonstration of effectiveness.
Complexity	The complexity of schemes is too great and not coordinated. For example, written instructions that are too long, too complex, or generally inaccessible. Overlap and incompatibility between schemes on the same farm
Mechanism	The mechanism or package for implementing different management measures needs to be considered carefully to ensure take-up, and will depend on the detail.

lems in their nearest watercourse. Many farmers are still not fully aware of the nutrient value of applied manure, or indeed the role of soil testing in deciding on fertilizer use, but both are management practices that may increase over some range of application, farm profitability by reducing dependence on fertilizers.

Sound information regarding the requirements for manure storage, and proper manure-handling techniques, must be promoted to minimize the potential for increased P loss. This is especially important for those livestock farmers operating on small land bases (<40 ha), where over-reliance on AFOs to supplement farm income leads to a decline in nutrient budgeting skills and inefficiencies of P use. In a recent study of farmer attitudes towards grant-aid in support of diffuse pollution control in England, there was a clear distinction between livestock and arable farmers (Withers et al., 2003). It was found that arable farmers are generally more skilled in nutrient and soil management than livestock farmers because the rate and timing of fertilizer inputs, and standard of cultivations, have a direct impact on crop yields and therefore profitability. Livestock farmers are better cushioned from poor land management decisions through the ability to compensate by importing extra feed to achieve profitability. This in turn exaggerates nutrient surpluses and the long-term risk of nutrient loss. Decisions on fertilizer inputs to land are also cushioned to some extent by the availability of farm manures.

Most field evaluations of BMP effectiveness at reducing watershed export of P conclude that nutrient management is the single most effective measure for controlling P losses (Sharpley and Rekolainen, 1997). This involves the use of regional soil testing programs that are flexible enough to accommodate differences among watersheds and development of manure management plans for confined animal operations (Sims, 2000). A possible shortcoming of education programs is that they may be inconsistent in recommendations and interpretation, especially where a number of different schemes with different objectives are operating on the same farm, for example, habitat conservation, soil protection, water quality controls etc. Advice given to control P loss may also conflict with advice given to control N loss (Dampney et al., 2003). For example in the UK, BMP for reducing manure P losses is to rapidly incorporate manures in the autumn when soil conditions are dry, but this practice is prohibited in Nitrate Vulnerable Zones due to the greatly increased risk of N leaching loss. Increased pesticide usage associated with establishing coarse seedbeds to reduce runoff risk, and increased sediment or N loss where cultivation is needed to incorporate manures to reduce runoff P losses are other examples (Dampney et al., 2003). Conflicting advice can lead farmers to question the reliability and philosophy of such programs, as well as a reluctance to use recommended management practices.

Success in education and extension also relates to social and economic factors. Education to control nonpoint source water pollutants is most effective when the action or management practice to improve water quality increases profitability, producers have strong stewardship motives, and/or large on-farm costs are associated with the pollution (Ribaudo and Horan, 1999).

Encouraging Farmer Involvement

One barrier to the development and implementation of sustainable management systems, is that system monitoring and assessment implemented by governments, is often perceived as a top-down process. Walker et al. (1996) recommend a bottom-up process for the successful selection of sustainable practices for use at a local level. A bottom-up process would seem equally if not more fundamental to the successful identification and adoption of new management systems. Withers et al. (2003a) similarly concluded that farmers preferred to be given a range of measures to choose from, and develop their own solutions, rather than be restricted to a particular BMP by a so-called expert who is not so familiar with the farming system or runoff risk. For example, a concerted attempt has been made in Australia to take this approach by devolving primary responsibility for local monitoring and resource management to land managers themselves through the provision of government funds for the national Landcare and Waterwatch programs. The national vision for the Decade of Landcare Plan is "the development and implementation of systems of land use and management which will sustain individual and community benefits now and in the future" (SCARM-ARMCANZ, 1997). In the farmer attitude survey reported by Withers et al. (2003a) for England, nearly all farmers were willing to take responsibility for controlling losses provided adequate training and advisory support was available. A number of LANDCARE initiatives based on this approach are now being organized in the UK following an initial pilot project by Huggins (1998).

In Pennsylvania, the Department of Environmental Protection, has committed to a watershed approach to manage water resources. Voluntary implementation of watershed management plans and nonpoint source pollution reduction strategies are left to local watershed residents (Dodd and Abdalla, 2002). Where agriculture is identified as a nonpoint source polluter, engaging and involving farmers in watershed management activities is fundamental to enhancing communication and cooperation. Watershed management in Pennsylvania can be improved by engaging all watershed stakeholders. Stakeholder involvement builds trust and support for the watershed approach, shares responsibilities for decisions or actions, creates solutions more likely to be adopted, leads to better, more cost-effective solutions, strengthens working relationships and enhances communication and coordination of resources (Tetra Tech, Inc., 1999).

Stakeholder alliances encourage collaborative, rather than adversarial, relationships among concerned parties. Such alliances have been formed in response to recent public health issues related to the nutrient enrichment of waters in the eastern USA. In the Chesapeake Bay, stakeholder alliances have developed among state, federal, and local groups and the public to work together to identify critical problems, focus resources, include watershed goals in planning, and implement effective strategies to safeguard soil and water resources (Chesapeake Bay Program, 1995, 1998). Similarly, in New York a Watershed Agriculture Council was formed of farmers, civic leaders, and representatives from the New York City Department of Environmental Protection to help guide management in the New York City Watershed (Revkin, 1995; also see http://www.nycwatershed.org/ [verified 6 Oct. 2004]).

PUBLIC POLICIES USING ECONOMIC INCENTIVES

Policy-makers and regulators can use a range of economic instruments to address nutrient related water quality concerns from agriculture. These mechanisms can be generally broken into: financial incentives (taxes and subsidies) and market-based approaches, such as trading.

Financial Incentives (Taxes and Subsidies)

One diagnosis of the nutrient problems resulting from agricultural production is that prices that emanate from food and agricultural markets as currently instituted within existing property rights are not "correct." That is, these prices do not reflect the true costs resulting from production. For example, the off-farm costs to fishermen or boaters of the eutrophication of surface waters from excess P loadings are not a factor that farmers must consider in their decisions. Yet these costs are very substantial. Pretty et al. (2003) calculated that freshwater eutrophication costs the UK £75 to 114 million per annum. An important implication is that prices do not provide the signals and incentives to market participants—from input suppliers to consumers to agricultural producers—that would reduce production or consumption of products that cause greater pollution and increase production or consumption of products that result in less pollution. The rationale for government intervention in the form of taxes or subsidies is to correct this "market failure" by changing the relative prices of inputs, activities, or products that cause environmental damage to market participants.

Several conceptual analyses have found that financial incentives (taxes or subsidies) have important desirable attributes compared to regulatory standards (Shortle and Dunn, 1991). In the nonpoint source water pollution context, two approaches have been identified for use of taxes: charges on inputs or practices (e.g., fertilizer and feed purchases or manure applications) and charges on proxies (e.g., nutrient applications in excess of plant needs or nutrient loadings from modeling efforts) for water quality emissions (Shortle and Horan, 2001). The taxation option has been studied by researchers in Europe and in the USA, but it has received little policy application as a way to change farmer behavior. Limited U.S. experience indicates that taxes are used as a means to raise government revenues, rather than change farmer behavior.

Subsidies (often called cost-sharing) have been a popular method in the USA for addressing soil erosion and water quality problems from farming. Public subsidies, like taxes, have the positive outcomes as a policy mechanism in that they provide farmers an incentive to seek production methods that are lower in cost to the producer and in environmental damage to society. However, there are several caveats to this conclusion. Subsidies should reduce environmental damage per farm, but use of this mechanism may not reduce overall pollution loadings in a watershed. The reason for this is that subsidies tend to discourage firm exits and may even encourage entry into a particular sector (Shortle and Dunn, 1991). Also, pragmatic issues may be encountered with the use of subsidies. Napier and Forster (1982) argued that sufficient public sector economic and social resources may not exist to address the great magnitude of soil and water problems resulting from agriculture in the USA. In England, it was estimated that advisory support was by far the biggest cost in implementing a subsidized control programs to control nonpoint sediment and P loss. There was also a general shortage in suitably skilled advisers to fulfil the advisory requirements, especially with regard to soil management and avoidance of soil compaction (Withers et al., 2003a). As a result, the UK Government are cautious and recommend that cost-sharing should be not be relied on solely as an approach to deal with farm-related erosion and water quality problems.

Dwyer et al. (2002) reviewed a number of schemes to tackle diffuse agricultural nutrient pollution in different countries (Table 32–6). Some countries adopt a more regulatory approach but without any real evidence that they are effective in improving water quality, even though reductions in fertilizer use may be achieved. Other countries adopt a mix of voluntary and subsidized or grant-aided approaches, with evidence that significant changes in farm practice can be achieved, especially where financial or environmental gain can be clearly established, and with farmer-owned planning and liaison with stakeholders, that is, a combined effort. Hence, in considering policy strategies relevant to England and

Table 32–6. Summary of Schemes to help control diffuse pollution in the European Union and USA.

Country	Scheme	Type	Scale	Management measures	Strengths	Weaknesses	Other
Netherlands	MINAS†	Regulatory	Farm-level country-wide (35% farms, 50% livestock farms). Coverage will increase in future.	Limits per ha for surplus N and P–levies applied if limits exceeded. Limits are lowered over time Mineral accounting compulsory. Levies charged on surplus manure.	No direct monitoring: modeling suggests significant pollution reduction: from 15 to 80% depending on modeling method used. Does not require extra paper work from farmers since accounts already 'standard practice' on these farms. Cost of compliance relatively low to date.	Significant administrative burden–Ministry now seeking alternative less rules- based approaches. Levies not high enough, do not affect enough farmers to achieve results; many farmers do not pay	Introduced in 1998–little data available. EFMA expects a significant reduction in use of fertilizer as a result of the scheme. From 2002 farmers producing excess manure have to secure contracts to transport it off farm or reduce cow numbers–unpopular.
Ireland	National P Reduction Strategy	Strategic	Local authority level Country-wide	Authorities had to submit plans detailing how they intended to meet targets set out in the regulations.	Addresses forestry and other sectors as well as agriculture–includes some direct restrictions.		Expected to be addressed via by-laws (see next box).
	Local Authority By-laws	Regulatory	Local authority level, most likely to implement: 16 plan to implement, 7 are still assessing need.	County councils can introduce requirements for nutrient management plans on all farms. REPS plans will qualify.	Can cover all farms, not just those in REPS. Considers pollution over larger area–for example, water abstraction a long way downstream.	Cannot prosecute on basis of no implementation. Cost of preparing plan must be met by the farmer.	Not yet implemented–likely to be established within next few years may encourage more farms into REPS.
	REPS	Grant aid and training	Farm-level, Country-wide. Currently agreements cover a third of total farmed area.	Farmers must follow farm nutrient plans (drawn up by certified planner), attend compulsory training to help them meet scheme requirements.	Popular with farmers (45 000 farms in) but new scheme less so (2000) than old (1993–1999). Improved water quality and P status.	Only includes farms that apply for participation. Does not take distance impacts into account.	Improvement in water quality first seen for 30 yr. Agriculture responsible for about 70% of nutrients entering waters.
Finland	GAEP	Grant aid	Country wide (applies to 80–90% farmland)	Whole farm scheme: 1.Environmental planning 2. Fertilizer limits 3. Plant protection rules 4. Conservation headlands and buffer strips 5. Maintenance of biodiversity and landscape. Additional measures with additional payments (over 5–10 yr).	Increased accuracy of fertilizer use, lower application rates shown. Modeling predicts significant N leaching reduction, reduced particulate P but increased dissolved P. Popular with farmers, cheap to run, high uptake means watershed scale effect.	No demonstration projects or training facilities are included in the scheme—farmers have to pay to receive advice to help them prepare plans, etc.	The scheme is generously funded and an important source of income support to farmers, so payments under the scheme merit farms taking external, paid advice when joining up.

France	CTE	Grant-aid and advice	Farm-level, country wide. Covers 3% of agricultural land in France; regional variation in uptake. Water quality objectives included in plans.	A range from a 'menu of options' are selected at Departmental level in conjunction with local farmers 55% of contracts involve measures to reduce water pollution.	Large menu of options tailored to local needs. Farmers obliged to implement environmental items to qualify for investment/diversification items. Promoted in some areas through group schemes.	Complex admin. Slow uptake due to learning curve requirement for local implementers. Management plans may be intensive (2–5 d advice). High workload for local technicians.	Too early to judge results RDP predicts that expenditure will be 4300 million Euro between 2000 to 2006.
	Ferti-Mieux	Advisory	Group action in local areas Priority watersheds for N pollution 49 groups established, covering 1.9 m ha	Nutrient management 'rational use' program using local advice, group discussion, and experiment to identify and apply better management practices. Label for groups who can demonstrate significant changes in farm practices–offers reduced regulatory costs, marketing potential.	Has reduced fertilizer use and area of soil left bare over winter. Real pollution reduction demonstrated in 2/3 project areas. Fairly cheap to run, £60 000 per area, from national and local partners funds and in-kind contributions.	Focused on nitrates to date, just beginning to tackle phosphates, soils, and pesticides.	Educational value: pioneering a new, more environmentally aware, partner orientated approach to agriculture. Emphasis on hearts and minds–farmer ownership, participation, and design.
USA	Conservation compliance	Cross compliance	High risk areas in priority watersheds (highly erodible land) 10% of U.S. cropland–all mapped under law.	Production subsidies conditional on compliance with environmental criteria–in particular, all farmers with highly erodible land have to have an approved soil conservation plan in place for this land.	Management measures generally low cost to farmers. Data suggests large reductions in soil erosion as result of the scheme. Nonmarket economic benefits $1400 million.	Only deals with soil erosion, no particular focus on nutrients.	Noncompliance very low (<5%), few cases prosecuted.

† MINAS = mineral accounting system, EFMA = European Fertilizer Manufacturers Association, REPS = Rural Environmental Protection Scheme, GAEP = General Agri-Environment Program, CTE = Contrats Territoriaux d'Exploitation.

Wales, the need to encourage 'bottom-up' farmer participation and the variable cost of the range in measures that might be adopted on- farm, these authors recommended a two-tier grant aid package to help control P loss. The first tier would provide financial support for basic nutrient and soil management planning with emphasis on adoption of nil, or low cost, measures that could be adopted country-wide, and would not adversely affect the farm business. The second tier would provide more generous payments for farmers to draw up more detailed farm plans, become more actively involved in watershed stakeholder partnerships and adopt more costly management measures. This second tier was aimed at priority watersheds with more pressing eutrophication problems and which needed more immediate action. Further piloting of this approach confirmed the central role of farm planning and indicated that such a package would be acceptable to the farming community provided sufficient advisory support and grants were available (Withers et al., 2003a). The costs of the advisory component to aid farm planning were greater than the costs of the measures which required to be implemented.

Specific Cost-share Programs

In the USA, there are numerous sources of technical assistance and financial cost-share and loan programs to help defray the costs of constructing or implementing practices that safeguard soil and water resources. Some of these sources are Conservation Technical Assistance (CTA), Conservation Reserve Program (CRP), Conservation Security Program (CSP), Environmental Quality Incentives Program (EQIP), Small Watershed Dam Restoration (SWDR), Special Water Quality Incentives (SWQI), Wetlands Reserve Program (WRP), and Wildlife Habitat Incentive Program (WHIP) (USDA, 2002).

The 2002 Farm Bill greatly expanded the amount of money available for cost-sharing subsidies through EQIP and other programs. The amount of federal funds available for EQIP program for the Fiscal Year 2002 to 2007 period will be \$5.8 billion, or more than 4.5 times the amount the total spent under the 1996 Farm Bill. Sixty percent of these funds will be used to address problems related to livestock, many of which are nutrient related. The issues related to whether this new program will reach water quality goals related include: whether the EQIP funds will be effectively targeted to problem areas and integrated well with other water quality programs (Clean Water Act's Total Maximum Daily Load program); and if sufficient public and private organizational resources will exist for effective implementation of the program (Abdalla and Dodd, 2002)

Watershed-based programs have been established to provide technical assistance and financial support to farmers participating in water quality protection programs. Perhaps the most prominent among these is the New York City Watershed Agriculture Program, where savings the city achieved through filtration avoidance has been used to subsidize farmer BMPs at up to 100% cost-share rates in upstate areas that feed that municipal drinking water reservoirs.

Market-based Approaches: Nutrient Trading

In order to develop nutrient management programs on a watershed scale, a system of buying and selling pollution credits within a given watershed, similar to that adopted for air quality control in the early 1990s, has been suggested as an approach for achieving water quality goals (USEPA, 2003). An important aspect of this approach is the need to establish limited "rights to pollute" of farmers or other watershed participants that would then allow the others in the "market" to trade with them. The appeal of trading is its potential to achieve water quality goals at a total lower cost to society than a regulatory approach. In a well-functioning market with trading, businesses with lower control costs will emit less pollution, while those with higher control costs will emit more. When considering all controls together, the total costs to society will be lower (Shortle and Horan, 2001).

The area that has been given most attention is potential trading between "point" and "nonpoint" sources of water pollution. While successes have been achieved in trading air quality permits, significant challenges have been encountered in applying this mechanism to water pollution. For instance, water quality emissions from nonpoint sources are difficult or impossible to observe and are related to uncertain events such as weather. Therefore, it is not clear what basis should be used to measure performance or outcomes (i.e., the "product" one buys or sells) from a trade. In addition, potential buyers and sellers to nonpoint source trades face significant costs of organizing and negotiating trades, called transaction costs. Relatively little study has been conducted on these types of costs as a barrier to water pollution trading (Shortle and Horan, 2001).

There continue to be many discussions among researchers and policymakers and research and demonstration efforts concerning trading among nonpoint and point water pollution sources. For example, the EPA is piloting programs in a number of states on nonpoint-point trading including some projects that involve farming (U.S. Environmental Protection Agency, 2003). Other developments include experimentation with new programs that involve possible trading among farmers. For example, farmers able to limit P loss below recommended levels could sell credits to a farmer unable to meet these levels. The number of credits a farmer has could be linked to farm area, crop production, and where appropriate number of animals. As a result, P export from a watershed may be kept within predetermined limits by sharing nutrient management responsibilities among farmers.

It should be noted, however, that "pollution trading" has been criticized by some environmental groups because it is perceived as allowing wealthier operations to buy the "right to pollute." Heated debate will likely precede the adoption of pollution credits for agriculture, hence comprehensive analysis and planning to justify their value and need will be required.

Despite these challenges, the lack of progress thus far with actual trades of water pollution permits, as well as environmental and others concerns noted above, there continues to be much effort to study and promote trading in the water pollution area. Two recent federal level developments related to trading are the water quality trading guidance (U.S. Environmental Protection Agency, 2003) and trading credit provisions in the proposed 2002 Farm Bill EQIP rules released (USDA, 2003).

Strategic Incentives

Along the lines of the discussion above, a strategic incentive approach also recognizes that market prices do not provide the signals or motivation to producers to manage nutrients in a way that minimizes environmental damage. In this approach, sometimes called consumer-based environmentalism, the key is to change the system so that consumers can make new choices about products that allow signals and extra revenues to be sent to producers for more environmentally friendly products. This assumes that consumers are interested and willing-to-pay for increased environmental protection and are able to distinguish and make product choices to express these preferences. Proponents of strategic incentives believe that for real and lasting changes to occur in agricultural production, emphasis must be placed on consumer-driven programs, as well as education rather than assuming that farmers will absorb the burden of pollution control costs. This is also the principle behind the Farm Assurance Schemes in England and current reforms of the Common Agricultural Policy (CAP) in Europe, where the subsidies the farmer receives is at least partly dependent on compliance with environmental conditions, which might include for example the preparation and adherence to a farm management plan which details sustainable nutrient, crop and soil management practices (Fig. 32–6).

New ways of using market signals and incentives to help farmers implement BMPs are needed. One example of an innovative consumer approach was tried in the northeast USA. A multi-agency collaborative venture, called the Dairy Network Partnership, released Chesapeake Milk in Fresh Fields Stores. For every half-gallon of Chesapeake Milk sold, 2.5 cents was returned to the certified Pennsylvania dairy farmers to reward their high environmental standards. Another 2.5 cents was deposited into an Environmental Quality Initiative (EQI) that provided a cost-share for those farmers who want to install conservation practices to qualify for the EQI program. While the approach produced some increased public awareness and generated some lessons about consumer-driven approaches, it did not prove to be self-sustaining effort (Lanyon, 2000).

Phosphorus Management Infrastructure

It must also be remembered that BMPs are "band-aids" to minimizing the impacts of agricultural P inputs on receiving waters. There is a need to look at strategic pressure points that control P inputs to farming systems and assess the need for planning of regional, as well as, national agricultural infrastructures (Lanyon, 2000). Infrastructure components might include education and extension programs. For example, cost share monies in northeast U.S. watersheds are now linked to AFOs showing that inputs are decreased by feeding animals at National Research Council requirements (National Research Council, 2001). This is an excellent example of addressing the source or cause of excess P concentrations and how public investments can provide a long-term mechanism for overcoming infrastructure barriers.

In geographic areas where livestock production is highly concentrated, there may be a case for investments in infrastructure that can collect, process, and redistribute manures to areas with high local demand for P. Delaware's Nutrient Management Relocation Program provides an example of public cost share monies reducing barriers to manure transportation. The program aims to reallocate manure from geographic areas with excess nutrients to areas in need of nutrients

(Delaware Department of Agriculture, 2004). While this innovative program has been successful at redistributing manure within the state of Delaware, it is important to note the program may negatively impact manure markets and distribution in surrounding states, such as Pennsylvania. Unintended consequences can occur when nutrient issues are addressed only within political or watershed boundaries. This suggests public investments in strategies and technologies for P source reduction as well as in reallocating manure are likely more effective at a regional or national levels.

Regulatory Approaches

It is clear that current technology and water quality legislation will not permit an unlimited number of animals in a region, or allow production of high-risk crops on land with a very high P loss risk. Thus, it may be necessary to limit animal numbers, or restrict cropping practices, within an area. Under CAP reforms in Europe, there is a proposal to revert all fields bordering watercourses (riparian land) to non-agricultural use because of the very high risk of pollution from land that has prolonged hydrological connectivity with the river, or flooding. Most U.S. states now require new animal facilities, which exceed a certain size, have an "approved" nutrient management plan. Clearly, it is essential that we develop and transfer technology to implement recommendations for the sustainable management of agricultural P, including recommending high-risk farming systems, or practices, be re-located to areas or fields with low P loss risk (Dwyer et al., 2002). For example, not growing continuous maize on the same field, but rotating this high-risk crop around the farm, avoiding high P loss risk fields.

In response to mounting water quality concerns, many states in the USA have developed guidelines for land application of P and watershed management based on the potential for P loss in agricultural runoff (USDA and USEPA, 1999). These actions have been spurred, in part, by a federal initiative in which the USDA and USEPA created a joint strategy to implement Comprehensive Nutrient Management Plans (CNMPs) on AFOs, with a national deadline of 2008. Under this strategy, USDA's Natural Resource Conservation Service (NRCS) is charged with implementing a new nutrient management policy. As a result, the NRCS planning standard that deals with nutrient management (590 standard), which was based on N, has been rewritten to include a P-management planning standard. In each state, NRCS State Conservationists must decide which of three P-management approaches will be used in nutrient management planning policy. These approaches are agronomic soil test P (STP) recommendations, environmental STP thresholds, or a P Index to rank fields according to their vulnerability to potential P loss.

A survey of the 50 states enacting CNMP strategies shows that 47 adopted the P Index approach, one adopted an agronomic STP (crop response) approach, and one adopted an environmental STP threshold approach (Fig. 32–7). The P Index was originally designed to assess the risk or vulnerability of P loss from a given agricultural field by accounting for source (soil P, fertilizer, and manure management) and transport factors (erosion, runoff, leaching, and connectivity to a stream channel) controlling P loss (Lemunyon and Gilbert, 1993; Gburek et al., 2000). The specific factors included in these Indices, how ratings are calculated,

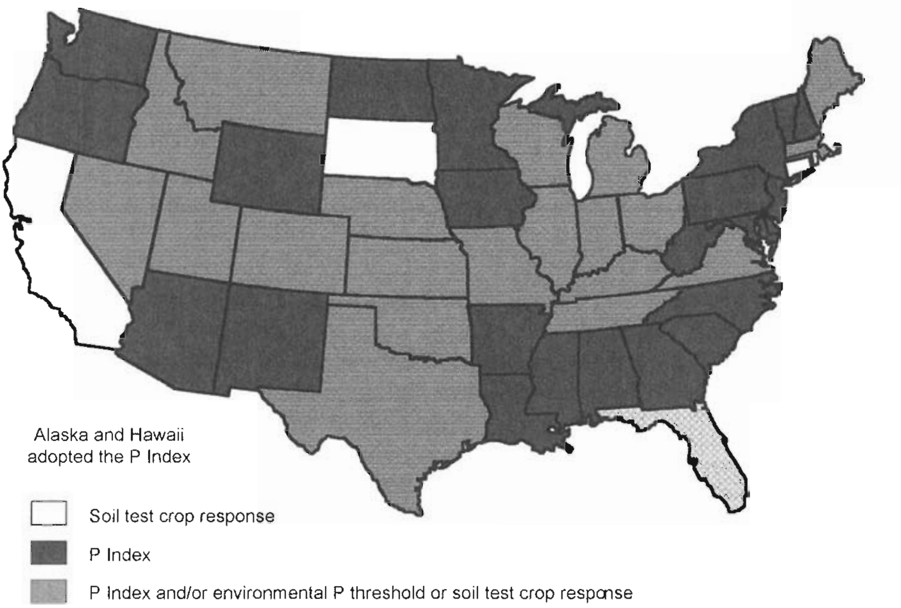

Fig. 32–7. Summary of phosphorus (P) management strategies adopted by state USDA-NRCS agencies for revision of the 590 nutrient management practice standard.

and whether the output is a risk or loss based estimate are given by Sharpley et al. (2003). Such widespread adoption of the P Index confirms a general scientific and policy consensus toward this approach as a valid, flexible means on which to base P-management recommendations for CNMPs.

While there has been broad adoption of the P Index concept, its development from a concept into a field assessment tool has followed several trends throughout the USA. The variations reflect not only regional differences in P movement, but philosophical differences as to how P risk from a site should be assessed using a P indexing approach. The fact that there are many modifications and versions of P Indices in use and accepted, demonstrates the robustness and flexibility of the P Indexing framework originally proposed by Lemunyon and Gilbert (1993). Also, within Europe, the P index approach has been promoted, developed and adapted (Heathwaite et al., 2003; Djodic et al, 2002) and forms the basis for the identification of high-risk areas and practices within UK modeling programs and in the development of operational guidelines for the safe use of P amendments

INTEGRATING SCIENCE INTO POLICY

Rather than conclude prima facie that inappropriate farm management is responsible for today's soil and water quality problems, we must also address the underlying reasons for the current situation. In many cases, the causes of current problems are related to marketplace pressures and the economic survival tactics of farmers. Sound research is needed to develop programs that encourage farmer performance and stewardship to achieve agreed production and environmental goals. These programs should focus on public participation to resolve conflicts between economic production efficiency and social issues such as water quality. Defensible, scientifically-based information and recommendations are essential if stakeholder and interest group misperceptions about proper environmental management are to be overcome. Fundamentally, the large gap in our understanding of the precise impacts of particulate and dissolved P loss on the chemical and biological response in different types of water bodies urgently requires filling. This information is needed to help justify the large investments required to promote and implement a program of sustainable P management across farm, watershed, regional and national scales.

In being information brokers between land managers and policy makers, scientists can help by giving more emphasis to the flow of information than has previously been the case. Increased emphasis in this area is essential for implementing management practices for better soil and water quality. Also, through this communication role, it is likely that many opportunities for research will arise, opportunities that will have a better chance of being realized because government and particularly industry ownership has been engendered through the communication process. These trends heighten the need for scientists to understand the practical, economic, and social issues from industry's point of view.

The gap between policy and science is currently being vigorously debated in the USA. For instance, new policies and programs that mandates implementation of nutrient management plans, particularly for agricultural systems involving animals, has required a shift in focus to address P as well as N (Lander et al., 1998; Sharpley et al., 1998). This shift has been driven by an increased incidence of freshwater eutrophication and toxic dinoflagellate outbreaks (Burkholder and Glasgow, 1997; Matuszak et al., 1997; U.S. Environmental Protection Agency, 1996b), even though there is no direct scientific evidence of a link between N or P and dinoflagellate outbreaks. Because of the shift from N to P management, some in the farming community feel misled by science and extension, which recommended N-based manure management to mitigate nitrate (NO_3^-) leaching to groundwater (Achenbach, 1998; Blankenship, 1997; Matuszeski, 1997). Associated soil P build up was often encouraged to enhance soil fertility. This policy has been fuelled by several misconceptions especially that soil is an infinite sink for P and erosion control will eliminate P loss from agricultural fields (Sharpley, 1996). To a certain extent, knowledgeable scientists must become more proactive in disseminating the latest information to a diverse community of soil and water resource users.

An example of research which might flow from involvement in industry-government communication is farm-scale research to evaluate all external and internal factors controlling nutrient balances and export to water bodies. Farmers are at the start of the "food chain" and their decisions are increasingly influenced by regional and often global economic pressures and constraints, over which they, and at times their industries, have little or no control. In the U.S. dairy industry since World War II, greater fertilizer N use which increased corn grain production and reduced costs, along with the promotion of a domestic soybean [*Glycine max* (L.) Merr.] processing industry, has dramatically increased the feed energy and protein available for enhanced animal productivity (Lanyon, 2000). Improved animal breeding, specialized feed concentrates, and new production technologies promoted greater animal productivity on a smaller land area. At the same time, the land base available for manure management has declined due to urban development, set aside land, reforestation, and the increase in concentrated animal production systems. As a result, animal farming has changed from land-based to capital or economically-driven systems. Thus, manure production and management issues facing farmers are to a large extent driven by external economic factors rather than environmental issues.

In 1999, the National Phosphorus Research Project (NPRP) was established in an attempt to provide information on sustainable P management and to bridge the gap between science and policy (Sharpley et al., 2002; see http://pswmru.arsup.psu.edu/phosphorus/nprp.htm [verified 6 Oct. 2004]). The NPRP represents a consortium of federal and state agencies, as well as land grant universities, with collaboration in more than 20 states. The Project was created to coordinate research across the USA to meet research, policy, and societal needs concerning agricultural P management. The major goals of the NPRP are to: (i) establish soil P thresholds in areas where P enrichment of waters (surface and subsurface) may impair water quality; (ii) develop a reliable indexing tool to identify landscapes that are vulnerable to P loss; and (iii) integrate this information into Comprehensive Nutrient Management Planning strategies at a watershed scale. To promote widespread relevance of NPRP's findings, a variety of linkages have been established. For instance, by working with university extension and agricultural consultants, the Project hopes to rapidly disseminate findings and receive feedback on its applicability at the farm level. In addition, NPRP coordinators are working closely with federal and state regulatory personnel so that future nutrient management strategies will be based on sound scientific information.

A broader information exchange group; the Southern Regional Extension and Research Activity—Information Exchange Group (SERA-IEG 17; see http://www.soil.ncsu.edu/sera17/ [verified 6 Oct. 2004]) "Minimizing agricultural phosphorus runoff losses for protection of the water resource," was established in the early 1990s, as an interagency task force to implement at a field level a workable procedure to assess site vulnerability to P loss in agricultural runoff. Since then, SERA-IEG 17 has synthesized and promoted sound scientific practices to meet user needs for managing natural resources to protect water quality and sustaining agricultural viability. A similar group has beene stablished under the E.U. Co-operation in Science and Technology' (COST) program to develop methodologies for more accurately assessing the agricultural contribution to Eutrophication (COST 832, http://www.cost832.alterra.nl [verified 6 Oct. 2004]). Within the UK, a specific P loss research program funded by the Department for Environment, Food and Rural Affairs has also been operating to provide a robust scientific base for policy decisions on reducing P loss from agricultural land to water (http://www.defra.gov.uk/environ/pollute/envpoll.htm [verified 6 Oct. 2004]). These groups provide national and international leadership on assessing and managing agricultural management impacts on soil and water resources, as well as fostering coordination and cooperation among Government action agencies, land grant universities and state soil and water conservation districts.

CONCLUSIONS

The evolution of sustainable P management is an ongoing process that is the responsibility of all involved; from farmers to consuming public to policy makers. For farmers, the transition to sustainable P management will in some cases, require incentives, or a combination of incentives and regulation. Even a decade ago, the National Research Council (1989) recommended Congress restructure federal commodity programs to remove disincentives for the adoption of sustainable techniques. It is now clear that corollary, incentives or rewards, can enhance the transition to sustainable P management (Lanyon, 2000; Sparrow et al., 2000). Even with these incentives, there is a critical need to instill in all participants that it is in everyone's best interests to make changes towards sustainability, however small, and that each can make a difference and contribute to advancing the entire system. Reaching the goal of sustainable P management is, without a doubt, the responsibility of all participants; each has its own part to play, its own unique contribution to make, and its own rewards.

REFERENCES

Abdalla, C.W. 2002. The industrialization of agriculture: Implications for public concern and environmental consequences of intensive livestock operations. Penn State J. Environ. Law 10 (2): 175–191.

Abdalla, C.W., and A. Dodd. 2002. 2002 Farm Bill: Three pressing issues in conservation. K. Bailey (ed.) Farm Bill Newsl. Fall 2002.

Abdalla, C.W., M. Edelman, H. Foster, D. Patton, A. Socolow, and B. Weber. 1997. Land use at the rural-urban fringe. Land use and rural-urban interface task force. Farm Foundation.

Achenbach, Jr., V. 1998. Nutrient management proposals challenge future of farming. Lancaster Farming 43:19. Saturday Mar. 14, 1998. Lancaster, PA.

Anderson, N.J. 1997. Reconstructing historical phosphorus concentrations in rural lakes using diatom models. p. 95–118. *In* H. Tunney et al. (ed.) P loss from soil to water. CAB Int. Press, Wallingford, Oxon, UK.

Blankenship, K. 1997. Many question just what is a 40% goal. Bay J. 7(7):1, 6. October 1997. Alliance for the Chesapeake Bay, Baltimore, MD.

Breimyer, H. F. 1962. The three economies of agriculture. J. Farm Econ. 44:679–699.

Burkholder, J.A., and H.B. Glasgow, Jr. 1997. *Pfiesteria piscicidia* and other Pfiesteria- dinoflagellates behaviors, impacts, and environmental controls. Limnol. Oceanogr. 42(5):1052–1075.

Carvalho, L., S. Maberly, L.May, C. Reynolds, M. Hughes, R. Brazier, L. Heathwaite, S.Liu, J. Hilton, D. Hornby. 2003. Risk assessment methodology for determining nutrient impacts in surface freshwater bodies. R&D Tech. Rep. P2-260/9. Environment Agency, Bristol, UK.

Chesapekake Bay Program. 1998. Federal agencies Chesapeake ecosystem unified plan. Chesapeake Bay Program, Annapolis, MD.

Cowell, S.J., and R. Clift. 1997. Impact assessment for LCAs involving agricultural production. Int. J. Life Cycl. Assess. 2 (2):99–103.

Critchley, C.N.R., B.J. Chambers, J.A. Fowbert, A. Bhogal, S.C. Rose, and R.A. Sanderson. 2002. Plant species richness, functional type and soil properties of grasslands and allied vegetation in English environmentally sensitive areas. Grass Forage Sci. 57:82–92.

Dampney, P.M.R., P. Mason, G. Goodlass, and J. Hillman. 2003. Methods and measures to minimise the diffuse pollution of water from agriculture–A critical appraisal. Final Rep. NT2507. Dep. for Environ., Food and Rural Affairs, Noble House, London, UK.

Delaware Department of Agriculture. 2004. Nutrient management program. Available at http://www.state.de.us/deptagri/nutrients/ (verified 6 Oct. 2004).

Department for Environment, Food and Rural Affairs. 2002. Code of good agricultural practice for the protection of water. Ministry of Agric., Fisheries and Food. MAFF Publ., London, UK.

Department for Environment, Food and Rural Affairs. 2003. Strategic review of diffuse water pollution from agriculture. Discussion Document. April 2003. Dep. for Environ., Food and Rural Affairs, Noble House, London, UK. Available at www.defra.gov.uk (verified 6 Oct. 2004).

Djodijk, F., H. Montas, A. Shirmohammadi, L. Bergstrøm, and B. Ulen. 2002. A decision support system for phosphorus management at a watershed scale. J. Environ. Qual. 31:937–945.

Dodd, A., and C. Abdalla. 2002. Watershed pollution budgets: Pennsylvania's new approach to restore polluted waters. Penn State Coop. Ext., Farm Economics. Issue no. 2. Publ. Distribution Ctr., Penn State Univ., University Park, PA.

Dwyer, J., R.Eaton, A. Farmer, D.Baldock, P.J.A. Withers, and P. Silcock. 2002. Policy mechanisms for the control of diffuse agricultural pollution with particular reference to grant aid. Res. Rep. 455. English Nature, Peterborough, UK.

European Commission. 2000. European Union Directive of the European Parliament and the Council 2000/60/EC. Establishing a Framework for Community Action in the Field of Water Policy. European Commission, Luxembourg.

Evans, R., L.C. Cuffman-Neff, and R. Nehring. 1996. Increases in agricultural productivity, 1948–1993. Updates on agricultural resources and environmental indicators. No. 6. USDA–Economic Res. Serv. U.S. Gov. Print. Office, Washington, DC.

Foy, R.H. 2005. The return of the phosphorus paradigm: Agricultural phosphorus and eutrophication. p. 911–939. *In* J.T. Sims and A.N. Sharpley (ed.) Phosphorus: Agriculture and the environment. Agron. Monogr. 46. ASA, CSSA, and SSSA, Madison, WI.

Gardner, G. 1998. Recycling organic wastes. p. 96–112. *In* L. Brown et al. (ed.) State of the world. W.W. Norton, New York.

Gaterell, M.R., R. Gay, R. Wilson, R.J. Gochin, and J.N. Lester. 2000. An economic and environmental evaluation of the opportunities for substituting phosphorus recovered from waste water treatment works in existing UK fertilizer markets. Environ. Tech. 21:1067–1084.

Gburek, W.J., A.N. Sharpley, L. Heathwaite, and G.J. Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. J. Environ. Qual. 29:130–144.

Gibson, G.R., R. Carlson, J. Simpson, E. Smeltzer, J. Gerritson, S. Chapra, S. Heiskary, J. Jones, and R. Kennedy. 2000. Nutrient criteria technical guidance manual: Lakes and reservoirs (EPA-822-B00-001). USEPA, Washington, DC. U.S. Gov. Print. Office, Washington, DC.

Gough, M.W., and R.H. Marrs. 1990. A comparison of soil fertility between semi-natural and agricultural plant communities: Implications for the creation of species-rich grassland on abandoned agricultural land. Biol. Conserv. 51:83–86.

Harwood, R.R. 1990. A history of sustainable agriculture. p. 3–19. *In* C.A. Edwards et al. (ed.) Sustainable agricultural systems. Soil and Water Conserv. Soc., Ankeny, IA.

Haygarth, P.M., P.J. Chapman, S.C. Jarvis, and R.V. Smith. 1998. Phosphorus budgets for two contrasting grassland farming systems in the UK. Soil Use Manage. 14:160–167.

Haygarth, P.M., and S.C. Jarvis. 1999. Transfer of phosphorus from agricultural soils. Adv. in Agron. 66:195–249.

Heathwaite, A.L., A.N. Sharpley, and M. Bechmann. 2003. Conceptual basis for a decision support framework to assess phosphorus loss at the field scale across Europe. J. Soil Plant Nutr. 166: 447–458.

Heiskary, S.A. 1989. Lake assessment program: A cooperative lake study program. Lake Reserv. Manage. 5:85–94.

Huggins, R. 1998. The Landcare project. p. 145–152. *In* T. Petchey et al. (ed.) Non-point pollution: II. The Scottish Agric. College, Aberdeen, UK.

Iserman, K. 1990. Share of agriculture in nitrogen and phosphorus emissions into the surface waters of Western Europe against the background of their eutrophication. Fert. Res. 26:253–269.

Johnes P.J, B. Moss, and G.L. Phillips. 1996. The determination of water quality by land use, livestock numbers and population data—Testing of a model for use in conservation and water quality management. Freshwater Biol. 36:451–473

Johnston, A.E., and I.R. Richards. 2003. Effectiveness of different precipitated phosphates as phosphorus sources for plants. Soil Use Manage. 19:45–49.

Kellogg, R.L., and C.H. Lander. 1999. Trends in the potential for nutrient loadings from confined livestock operations. *In* The state of North America's private land. USDA-NRCS, U.S. Gov. Print. Office, Washington, DC. Available at www.nrcs.usda.gov/technical/land/pubs/ntrend.html (verified 20 Oct. 2004).

Lander, C.H., D. Moffitt, and K. Alt. 1998. Nutrients available from livestock manure relative to crop growth requirements. Resource Assessment and Strategic Planning Working Paper 98-1. USDA, Natural Resources Conserv. Serv., Washington, DC. Available at http://www.nhq.nrcs.usda.gov/land/pubs/nlweb.html (verified).

Lanyon, L.E.. 1995. Does nitrogen cycle? Changes in the spatial dynamics of nitrogen with industrial nitrogen fixation. J. Prod. Agric. 8:70–78.

Lanyon, L.E. 2000. Nutrient management: Regional issues affecting the Chesapeake Bay. p. 145–158. *In* A.N. Sharpley (ed.) Agriculture and phosphorus management: The Chesapeake Bay. CRC Press, Boca Raton, FL.

Lemunyon, J.L., and R.G. Gilbert. 1993. The concept and need for a phosphorus assessment tool. J. Prod. Agric. 6:483–496.

Magnien, R., D. Boward, and S. Bieber. 1995. The State of the Chesapeake Bay. U.S. Gov. Print. Office, Washington, DC.

Mainstone C.P., and W. Parr. 2002. Phosphorus in rivers–Ecology and management. Sci. Total Environ. 282–283: 25–47.

Marrs, R.H., and M.W. Gough. 1989. Soil fertility: A potential problem for habitat restoration. p. 29–44. *In* G.P. Buckley (ed.) Biological habitat restoration. Bellhaven Press, London, UK.

Matuszeski, W. 1997. Pfiesteria forcing us to rethink handling of nutrients in the Bay. Bay J. 7(7): 16–17.

Matuszak, D.L., M. Sanders, J.L. Taylor, and M.P. Wasserman. 1997. Toxic *Pfiesteria* and human health. Maryland Med. J. 46:515–520.

McCollum, R.E. 1991. Buildup and decline in soil phosphorus: 30-year trends on a Typic Umprabuult. Agron. J. 83:77–85.

McDowell, R.W., A.N. Sharpley, and A.T. Chalmers. 2002. Land use and flow regime effects in phosphorus chemical dynamics in the fluvial sediment o the Winooski River, Vermont. Ecol. Eng. 18: 477–487.

Napier, T.L., and D.L. Forster. 1982. Farmer attitudes and behavior associated with soil erosion control. *In* H.G Halcrow et al. (ed.) Soil conservation policies, institutions, and incentives. Soil and Water Conserv. Soc., Ankeny, IA.

National Research Council. 1989. Report on alternative agriculture. Natl. Academic Press, Washington, DC.

National Research Council. 1993. Soil and water quality: An agenda for agriculture. Natl. Academic Press, Washington, DC.

National Research Council. 2001. Nutrient requirements of dairy cattle. 7th rev. ed. Natl. Academy Sci., Washington, DC.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Annals Assoc. Am. Geogr. 77:118–125.

Organisation of Economic Cooperation and Development. 1982. Eutrophication of waters: Monitoring, assessment and control. Organisation of Econ. Coop. and Development, Paris, France.

Pagano, A.P., and C.W. Abdalla. 1994. Clustering in animal agriculture: Economic trends and policy. p. 192–199. *In* D.E. Storm and K.G. Casey (ed.) Proc. of Balancing Animal Production and the Environment. Publ.151. Great Plains Agric. Council. Kansas State Univ., Manhattan.

Pierson, S.T., M.L. Cabrera, G.K. Evanylo, H.A. Kuykendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784–1789.

Pionke, H.B., W.J. Gburek, A.N. Sharpley, and R.R. Schnabel. 1996. Flow and nutrient export patterns for an agricultural hill-land watershed. Water Resour. Res. 32:1795–1804.

Poulsen, H.D. 2000. Phosphorus utilization and excretion in pig production. J. Environ. Qual. 29: 24–27.

Pretty, J.N., C.F. Mason, D.B. Nedwell, R.E. Hine, S. Leaf, and R. Dils. 2003. The environmental costs of fresh water eutrophication in England and Wales. Environ. Sci. Technol. 37 (2):201–208.

Revkin, A.C. 1995. Unusual partnership farmers help safeguard New York water. New York Times. p. 1, 36, Aug. 13, 1995.

Ribaudo, M.A., and R.D. Horan. 1999. The role of education in nonpoint source pollution control policy. Rev. Agric. Econ. 21(2):331–343.

Schoumans, O.F., and W.J. Chardon. 2003. Risk assessment methodologies for predicting phosphorus losses. J. Plant. Nutr. Soil Sci. 166:403–408.

Sharpley, A.N. 1996. Myths about phosphorus. p. 60–75. *In* M. Salis and J. Popow (ed.) Animal agriculture and the environment: Nutrients, pathogens, and community relations. Northeast Regional Agric. Eng. Serv., NREAS-96, Ithaca, NY.

Sharpley, A.N. (ed.) 2000. Agriculture and phosphorus management: The Chesapeake Bay. CRC Press, Boca Raton, FL.

Sharpley, A.N., P.J.A. Kleinman, R.J. Wright, T.C. Daniel, B.C. Joern, R. Parry, and T. Sobecki. 2002. The National Phosphorus Project: Addressing the interface of agriculture and environmental phosphorus management. p. 95–101. *In* J. Steenvoorden et al. (ed.) Agricultural effects on ground and surface waters: Research at the edge of science and society. Int. Assoc. of Hydrol. Sci., Publ. 273. IAHS Press, Wallingford, Oxon, UK.

Sharpley, A.N., J.J. Meisinger, A. Breeuwsma, J.T. Sims, T.C. Daniel, and J.S. Schepers. 1998. Impacts of animal manure management on ground and surface water quality. p. 173–242. *In* J.L. Hatfield and B.A. Stewart (ed.) Animal waste utilization: Effective use of manure as a soil resource. Sleeping Bear Press, Boca Raton, FL.

Sharpley, A.N., and S. Rekolainen. 1997. P in agriculture and its environmental implications. p. 1–54. *In* H. Tunney et al. (ed.) P loss from soil to water. CAB Int. Press, Wallingford, Oxon, UK.

Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.L. Gburek, P.A. Moore, and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the U.S. J. Soil Water Conserv. 58:137–152.

Sharpley, A.N., and P.J.A. Withers. 1994. The environmentally sound management of agricultural phosphorus. Fert. Res. 39:133–146.

Shortle, J.S., and J.W. Dunn. 1991. The economics of control of nonpoint pollution from agriculture. p. 29–47. *In* N.D. Hanley (ed.) Farming and the countryside: An economic analysis of external costs and benefits. CAB Int., Wallingford, Oxon, UK.

Shortle, J.S., and R.D. Horan, 2001. The economics of nonpoint pollution control. J. Econ. Surv. 15: 256–289.

Sileika, A.S., S. Kutra,, and L. Berankiene. 2002. Phosphate runoff in the Nevezis river (Lithuania). Environ. Monitor. Assess. 78:153–167.

Sims, J.T. 2000. The role of soil testing in environmental risk assessment for phosphorus. p. 57–82. *In* A.N. Sharpley (ed.) Agriculture and phosphorus management: The Chesapeake Bay. CRC Press, Boca Raton, FL.

Sims, J.T., and P.J.A. Kleinman. 2005. Managing agricultural phosphorus for environmental protection. p. 1021–1068. *In* J.T. Sims and A.N. Sharpley (ed.) Phosphorus: Agriculture and the environment. ASA, CSSA, and SSSA, Madison, WI.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. P losses in agricultural drainage: Historical perspective and current research. J. Environ. Qual. 27:277–293.

Sparrow, L.A., A.N. Sharpley, and D.J. Reuter. 2000. Safeguarding soil and water quality. *In* Opportunities for the 21st Century: Expanding the horizons for soil, plant, and water analysis. Commun. Soil Sci. Plant Anal. 31:1717–1742.

Standing Committee on Agriculture and Resource Management-Agriculture and Resource Management Council of Australia and New Zealand. 1997. Evaluation report on the decade of landcare plan. Natl. Overview. Australian Gov. Publ. Serv., Canberra, Australia.

Tetra Tech, Inc. 1999. Getting in step: Engaging and involving stakeholders in your watershed. Guidebook prepared under contract 68-C-99-249 to the U.S. Environmental Protection Agency.

U.S. Department of Agriculture. 2002. Farm Bill 2002. Available at http://www.usda.gov/farmbill/ (verified 6 Oct. 2004).

U.S. Department of Agriculture. 2003. 2002 Farm Bill proposed EQIP rules, January 2003. Available at http://www.nrcs.usda.gov/programs/Env_Assess/EQIP/EQIP_PR.pdf (verified 20 Oct. 2004).

U.S. Department of Agriculture and U.S. Environmental Protection Agency. 1999. Unified national strategy for Animal Feeding Operations. March 9, 1999. U.S. Gov. Print. Office, Washington, DC.

U.S. Environmental Protection Agency. 1996a. Level III ecoregions of the continental United States. U.S. Environmental Protection Agency, National Health and Environmental Effects Res. Lab., Corvallis, OR.

U.S. Environmental Protection Agency. 1996b. Environmental indicators of water quality in the United States. EPA 841-R-96-002. USEPA, Office of Water (4503F), U.S. Gov. Print. Office, Washington, DC.

U.S. Environmental Protection Agency. 1998. National strategy for the development of regional nutrient criteria. EPA-822-F-98-002. USEPA, Office of Water (4304), U.S. Gov. Print. Office, Washington, DC. Available at http://www.epa.gov/waterscience/standards/nutsi.html (verified 6 Oct. 2004).

U.S. Environmental Protection Agency. 2001a. Ecoregional nutrient criteria. EPA-822-F-01-010. USEPA, Office of Water (4304), U.S. Gov. Print. Office, Washington, DC. Available at http://www.epa.gov/waterscience/criteria/nutrient/9docfs.pdf (verified 6 Oct. 2004).

U.S. Environmental Protection Agency. 2001b. Prepared by the Office of Wastewater Management under the direction of Mr. Gregory Beatty and Mr. Hank Zygmunt. Technical assistance was provided by Tetra Tech, Inc. under EPA Contract No. 68-C-99-253. Alternative Technologies/Uses For Manure. Available at http://www.epa.gov/npdes/pubs/cafo_report.pdf (verified 20 Oct. 2004).

U.S. Environmental Protection Agency. 2003. Water quality trading policy. 2003. Available at http: //www.epa.gov/owow/watershed/trading.htm (verified 6 Oct. 2004).

U.S. Food, Agriculture, Conservation, and Trade Act. 1990. Public Law 101-624, Title XVI, Subtitle A, Section 1603. U.S. Gov. Print. Office, Washington, DC.

U.S. Geological Survey. 1999. The quality of our nation's waters: Nutrients and pesticides. U.S. Geol. Surv. Circ. 1225. USGS Information Serv., Denver, CO. Available at http://www.usgs.gov (verified 6 Oct. 2004).

Valk, H., J.A. Metcalf, and P.J.A. Withers. 2000. Prospects for minimizing phosphorus-excretion in ruminants by dietary manipulation. J. Environ. Qual. 29:28–36.

Vighi, M., and G. Chiaudani. 1985. A simple method to estimate lake phosphorus concentrations resulting from natural loadings. Water Res. 19:987–991

Walker, J., D. Alexander, C. Irons, B. Jones, H. Penridge, and D. Rapport. 1996. Watershed health indicators: An overview. p. 3–18. *In* J. Walker and D.J. Reuter (ed.) Indicators of Watershed Health: A technical perspective. CSIRO, Collingwood, Victoria, Australia.

Withers, P.J.A., I.A. Davidson, and R.H. Foy. 2000. Prospects for controlling diffuse phosphorus loss to water. J. Environ. Qual. 29:167–175.

Withers, P.J.A., A.C. Edwards, and R.H. Foy. 2002. Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. Soil Use Manage. 17:139–149.

Withers, P.J.A., and S.C. Jarvis. 1998. Mitigation options for diffuse phosphorus loss to water. Soil Use Manage. 14:186–192.

Withers, P.J.A., and E.L. Lord. 2002. Agricultural nutrient inputs to rivers and groundwaters in the UK: Policy environmental management and research needs.. Soil Use Manage. 14:186–192.

Withers, P.J.A., S. Royle, M. Tucker, R. Watson, T. Scott, P. Silcock and J. Dwyer. 2003a. Field development of grant aid proposals for the control of diffuse agricultural pollution. R&D Tech. Rep. P2-261/09/TR. Environment Agency, Bristol, UK.

Withers, P.J.A., B. Ulen, C. Stamm, and M. Bechmann. 2003b. Incidental phosphorus losses—Are they significant and can they be predicted? J. Plant Nutr. Soil Sci. 166:459–468.

Wu, Z., and L.D. Satter. 2000. Milk production and reproductive performance of dairy cows fed two concentrations of phosphorus for two years. J. Dairy Sci. 83:1052–1063.