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The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation

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Abstract

The magnitude of current nutrient losses from agriculture to ground and surface water calls for effective environmental policy, including the use of regulation. Nutrient loss is experienced in many countries despite differences in the organisation and intensity of agricultural production. However, at present there is no internationally agreed practice to assess the effectiveness of different kinds of regulatory practice and compliance level, or to make effective comparisons. There is a wide variety of indicators available for this purpose, ranging from livestock density and input–ouput balances to nutrient concentrations in soil and water. This paper explores the effectiveness and efficiency of the different indicators, both in terms of achieving a single objective and a comprehensive set of objectives and evaluates how responsive and attributable these indicators are to changes in farm management.

Each indicator appears to have its own pros and cons. Unfortunately, there does not seem to be a single indicator that is effective, comprehensive, efficient, responsive and attributable at the same time. Scientifically, there are valid reasons to develop indicators that account for specific regional conditions and to accept the use of varying threshold values within them. Administratively complex indicators and/or strict threshold values appear inevitable wherever intensive production is the dominant form of land use amidst vulnerable local environments. However, at both a national and international level this differentiation of indicators and thresholds may conflict with the desire to treat individual farmers equally and to minimise their administrative burden. The paper concludes with a review of the issues raised in international harmonisation of nutrient loss indicators and threshold values.

Keywords: Agriculture; Environment; Nutrient loss; Harmonisation; Indicator; Legislation; Manure; Nitrogen; Phosphorus; Water quality

1. Introduction

Nitrogen (N) and phosphorus (P) are indispensable inputs for the sustainability of agriculture. The use of both inputs has increased dramatically in recent decades, but so have the nutrient losses, especially as N cannot be fully utilised in any production system (Isermann, 1993; Galloway, 1998; Carton and Jarvis, 2001; De Clercq et al., 2001b; Neeteson et al., 2003). Nutrient losses have a number of environmental consequences. N and P losses in particular can negatively affect the quality of soils, ground water, surface water, and the atmosphere. The losses may put drinking water quality at risk. They may also affect the functioning of ecosystems, including the earth as a whole (IPCC, 1996; Galloway, 1998; Tunney et al., 1997; Pierzynski et al., 2000; Carton and Jarvis, 2001; Freibauer, 2003). The financial consequences of these losses on a societal scale are considerable as well (Pretty et al., 2003). Thus, at the same time as their positive and self-evident production effects, N and P inputs to agriculture can also have detrimental effects on the health and welfare of present and future generations.

Agriculture has been found to be a major contributor to N and P losses to the environment (Cartwright et al., 1991; Novotny, 1999; Pretty et al., 2003), which justifies the call for effective environmental policy, whether this takes the form of economic incentives to stimulate benign nutrient management practices or (prohibitive) fees on undesired practices or outputs. Choosing the most effective policy mix of economic incentives and regulation is a complicated issue, not in the least, because action (at the farm management level) and response (the environmental effect) do not normally coincide in space and time (Menzel, 1991; Novotny, 1999; Pretty et al., 2003).

To evaluate whether the policy approach chosen is effective in reducing the N and P losses from agriculture, scientifically sound indicators must be defined and monitored. Various indicators have been proposed (De Clercq

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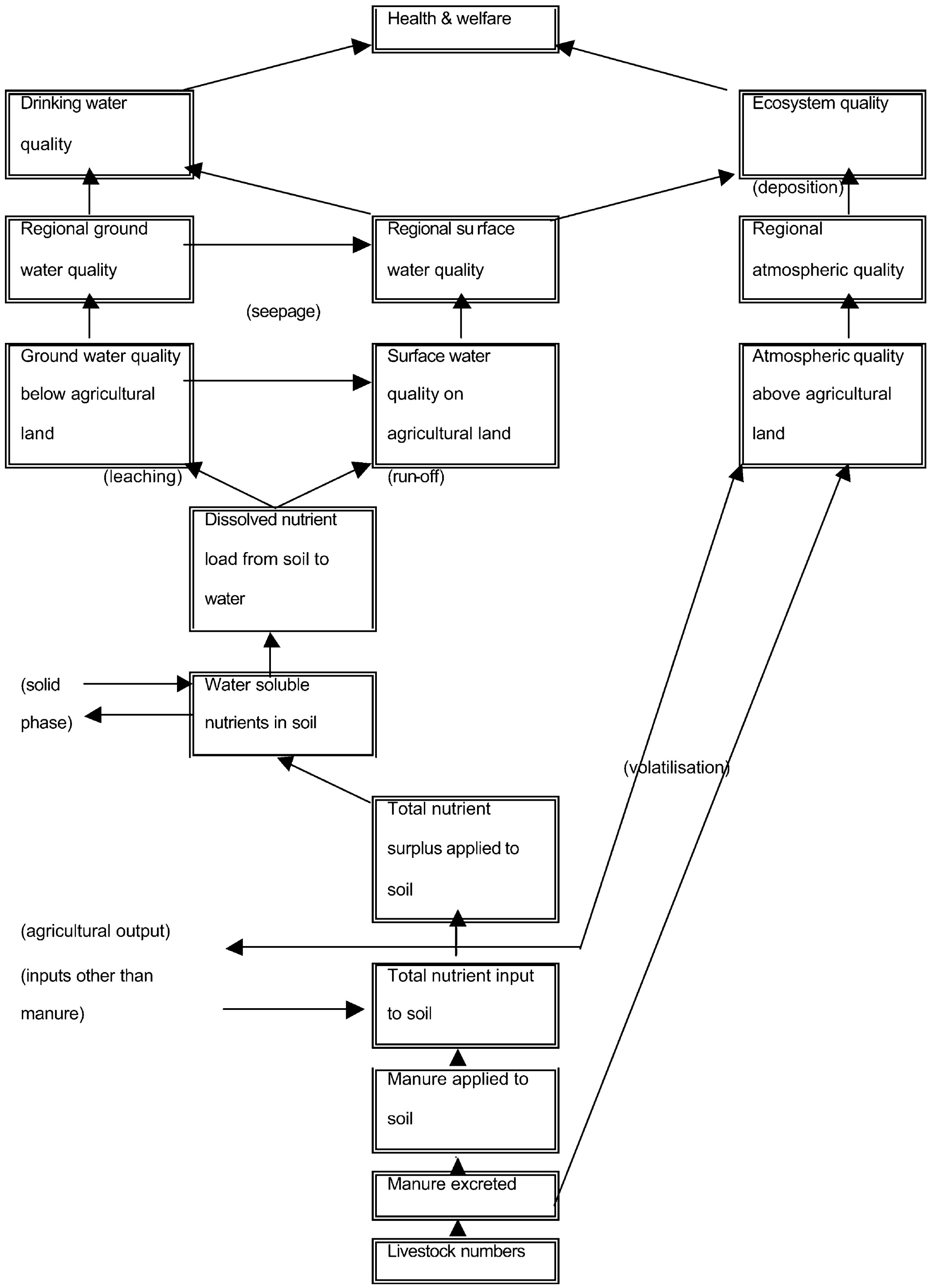

Fig. 1. Hierarchy of indicators to assess and control the impact of nutrient loss from agriculture on the quality of drinking water and ecosystems.

et al., 2001b; Van der Werf and Petit, 2002) and can be put into a hierarchical order which illustrates their interrelation (Fig. 1).

When linked to a threshold value and a monitoring system, indicators may be used in the setting of legislative standards or in a system of economic incentives to allocate premiums or fees to farmers. The indicators given in Fig. 1 may be defined according to various spatial and temporal scales, reflecting the allowance for averaging. The scale at which each indicator is monitored (e.g. field or region, daily or yearly) is critical as it determines to what extent peaks in losses can be compensated for by locations or time periods with lower losses. Farmers can be expected to follow the selection of indicators and definition of threshold values very critically, as both can have drastic consequences on their current nutrient management practices.

Scientific research has shown that there is a great deal of noise in the relationships between the various indicators, implying that achievement of the ultimate goal, i.e. a targeted state of health and welfare (in as far as these are determined by N and P losses from agriculture), becomes less certain the further an indicator is positioned from the ultimate goal.

This paper focuses on N and P losses from agriculture to ground and surface water in relation to the Nitrate (Anonymous, 1991) and Water Framework (Anonymous, 2000) Directives of the European Union (EU). The paper aims (1) to assess the effectiveness of indicators in relation to sources of variation, (2) to present criteria on how to select indicators and how to define threshold values for N and P losses, and (3) to explore the necessity and possibilities for harmonisation in the selection of indicators and thresholds.

2. Sources of variation affecting relationships between indicators

2.1. The impact of drinking water quality and ecosystem quality on human health and welfare

Health and welfare are broad societal goals and hence are hard to quantify. Societies differ in their evaluation of the extent to which water quality determines health and welfare. Thus, the political appreciation of the immediate and nearby availability of good drinking water or well-functioning ecosystems differs among nations due to arbitrarily set societal goals. However, most nations agree that nitrate-N in drinking water can negatively affect human health. Therefore, the World Health Organisation has set a standard limit for the nitrate concentration allowed in drinking water (50 mg nitrate/l) (Anonymous, 1993), which has recently been reaffirmed by the EU (Anonymous, 1998). Standards are meant to provide a high degree of protection to the consumer against exposition to toxic substances. In the case of nitrate, the current maximum level is not considered harmful to the consumer and an incidental exceedance of the acceptable daily intake will only cause negligible risks. However, the level at which nitrate becomes toxic remains a matter of harsh debate between scientists advocating the precautionary principle and scientists challenging the epidemiological evidence for direct relationships between nitrate and health (Boink and Speijers, 2001; L'Hirondel and L'Hirondel, 2002).

Health and welfare are also determined by the quality of local and global ecosystems, in which N and P losses from agriculture, among other factors, do play a major role via both direct and indirect effects. Direct effects pertain to air and water-borne eutrophication and the resulting general loss of biodiversity and blooming of toxic algae (e.g. Menzel, 1991; Pretty et al., 2003). Indirectly, N loss is associated with acidification via atmospheric deposition and with denitrification-induced solution of sulphate and metals, destruction of ozone and global warming (IPCC, 1996; Galloway, 1998; Tunney et al., 1997; Pierzynski et al., 2000; Carton and Jarvis, 2001; Freibauer, 2003). Moreover, the production and transport of fertiliser required to compensate for these losses has a detrimental effect on the environment as well, due to its large energy consumption (Corré et al., 2003). The relative contribution of the many factors determining ecosystem functioning is much debated, but N and P losses from agriculture are strongly believed to play a major role (Cartwright et al., 1991; Menzel, 1991; Galloway, 1998; Tunney et al., 1997; Novotny, 1999; Pierzynski et al., 2000; Carton and Jarvis, 2001; Pretty et al., 2003).

2.2. Regional ground and surface water quality as a function of water quality below and on agricultural land

Regional water quality in terms of N and P concentrations, including that in coastal zones, is not just determined by the quality of water below and on agricultural land, but also by the discharge of water, N and P from land uses other than agriculture. At this point, it should be mentioned that the nutrient load from agriculture is at least temporarily masked due to chemical processes in its runoff to downstream water systems. The spatial scale at which one wishes objectives to be achieved, strongly determines to what extent 'dilution' from these non-agricultural areas can be taken into account. Obviously, the impact of agriculture on regional water quality will become more evident when agriculture is the dominant form of land use. What may initially look like a negligible diffuse loss from the field of an individual farm, may eventually have serious ecological consequences downstream where the effects of numerous small diffuse sources accumulate (Burt and Haycock, 1991).

As far as biodiversity is concerned, one should also bear in mind that each group of species may have specific demands concerning N and P concentrations in water. Thus, the species for which regulations are targeted will also determine which losses are permissible. These are the considerations (watershed scale, ecological targets) which the recent EU Water Framework Directive (Anonymous, 2000) intends to take into account.

2.3. Ground water quality below and surface water quality on agricultural land as a function of water-soluble nutrients in the soil

Water quality in the direct vicinity of agricultural land can be related to the soil supply of water-soluble nutrients. Chardon and Van Faasen (1999) concluded that P-loss to ground and surface water has a strong positive relationship with the amount of water-soluble P in the topsoil. Positive relationships have also been established between the amount of residual soil mineral N (RSMN) at harvest and the nitrate concentration of the upper groundwater (Prins et al., 1998; Neeteson, 1994; Schröder et al., 1996b), although it must be emphasised that both relationships may be strongly variable. In the first place, nutrients are not only lost after harvest or in the form of water-soluble components. Secondly, losses due to leaching, seepage and surface run-off may differ. Hence, residence time in the soil and consequently, the probability of chemical conversion is likely to differ. Chemical conversions also depend on pedo-climatic factors, as illustrated by the considerable differences in denitrification (e.g. Nieder et al., 1989; Addiscott and Powlson, 1992; Aulakh et al., 1992) and P-fixation (e.g. Van der Zee et al., 1990a,b; Anonymous, 1997; Pierzynski et al., 2000) between different soils. However, soils do not have an infinite capacity to sequester (N, P) or reduce (e.g. NO_3–N) nutrients. Besides, denitrification is not as harmless as was once thought, as nitrous oxides are an inevitable by-product in addition to the production of innocuous elementary N (e.g. Nieder et al., 1989; Aulakh et al., 1992; IPCC, 1996; Velthof and Oenema, 1997; Galloway, 1998; Velthof et al., 1998).

The fact that a large number of factors are involved in determining water quality explains why standardised protocols on where, when and how to measure, are extremely important for sound evaluation and comparisons between different locations and over time. It does make a difference, for instance, whether the concentration of water-soluble P refers to just the upper 0–30 cm or to the 0–90 cm layer. The concentration of nitrate, too, is quite sensitive to sampling depth.

The eventual nutrient concentration in leachate, seepage or runoff is determined by the precipitation surplus as well. Low precipitation surpluses permit only small losses to stay below required concentration levels. If one aims for effective regulation, threshold values linked to the water-soluble nutrient supply in the soil will need to be differentiated for the many modifying factors involved, including precipitation.

Despite these many sources of variation, RSMN in particular, has been proposed as an indicator of N-emission to groundwater (Prins et al., 1998; Neeteson, 1994; Schröder et al., 1996b, 2000). Van Dijk (1991) and Corré (1994) argue that this indicator has an questionable 'snap shot' character, as RSMN may change in time due to C/N-driven mineralisation–immobilisation turnover of, e.g. crop residues (De Neve and Hofman, 1998), the modifying effect of summer rainfall (Schröder et al., 2000), variation due to soil sampling procedures (Schröder et al., 2000), post-harvest management (Schröder et al., 1996a, 1997; Silgram and Shepherd, 1997; Stenberg et al., 1999), and the heterogeneity of a field (Van Meirvenne and Hofman, 1989; De Willigen et al., 1992). Nevertheless, RSMN is currently being used as an indicator in the German country Baden-Würtenberg (Happe et al., 2001), in Flanders (De Clercq et al., 2001a) and on an experimental scale in The Netherlands (Ten Berge, 2002).

2.4. *Water-soluble nutrients in the soil as a function of the nutrient surplus*

Nutrient balance sheets also represent an indicator of nutrient losses, at least in the long run (Janssen, 1999; Van Noordwijk, 1999). Balance sheets consist of listed inputs and outputs and their difference or surplus, resulting in soil surface balances and farm gate balances (Watson and Atkinson, 1999; Schröder et al., 2003). The soil surface balance lists all ingoing and outgoing flows at the field level and is mainly directed at assessment of the N or P input into the soil. Nutrients in manure, mineral fertilisers, atmospheric deposition and from biological N-fixation are the major inputs and nutrients removed by crop products are the major output. In a farm gate balance, manure (unless imported to or exported from the farm) and forage crops (unless exported) are considered as internal flows. Fertilisers, biologically fixed N, atmospheric deposition and feed imports and agricultural produce exports are the main balance terms in such a farm gate balance.

Most of the terms in either the soil surface or farm gate balances are of an unambiguous nature: they are either input or output. The position of soil pool changes, by contrast, is not always so obvious. In a long-term steady state situation, for instance, mineralisation and immobilisation (which in a broad sense both determine changes of the soil N and P pools) are in equilibrium, at least at the scale of the rotation as a whole. Hence, their net change is zero so these terms can be omitted. At the scale of individual fields and in dynamic farming systems such as 'shifting cultivation', however, this assumption may not apply on an annual basis but only in the long run. In soils originating from recently reclaimed marshes and polders, in drained peat soils, and in soils in transition from a high input to a low input system (Motavalli et al., 1992; Whitmore and Schröder, 1996), mineralisation will generally exceed immobilisation. P-saturated soils can remain leaky for many years, long after outputs have been balanced by inputs, whereas P-fixing soils represent situations in which a positive surplus does not instantaneously lead to increased levels of water-soluble P in the soil and, hence, an immediate risk to the environment (Tunney et al., 1997; Pierzynski et al., 2000).

Similarly, a large N surplus may overestimate the risk of losses in systems that have recently resumed manuring. In such a situation, the evolution of mineral N from organic inputs is likely to be initially smaller than the full annual input (Schröder, 2002). Under those circumstances, the amount of RSMN resulting from a certain annual surplus is initially lower in systems based on manure than in systems based on mineral fertilisers (Wijnands and Van Leeuwen-Haagsma, 1997).

The N surplus of a farm gate balance expresses all potential losses to the environment, including gaseous losses of which ammonia–N is important in terms of kg N. Similar surpluses may be associated with completely different loads of the soil, as ammonia volatilisation varies strongly with the type of housing, manure storage, manure application techniques, the composition of animal diets and the extent to which animals are allowed to graze (Bussink and Oenema, 1998; Monteny and Erisman, 1998). So, if the surplus of a farm gate balance is intended to indicate risks of N losses from the soil only (i.e. denitrification, leaching, runoff, seepage), ammonia volatilisation should be transferred to the output side of the equation.

Balance sheet calculations are generally relatively blunt, given their inability to separately quantify different types of loss, even when the surplus has been corrected for losses of ammonia–N. For that purpose several indicators would be needed, one for each separate item. At the same time one could say that balance sheets have an integrative character in encompassing and thus controlling the various kinds of losses simultaneously.

The previous section shows that a balance sheet is not a perfect indicator per se of the N and P supply at direct risk to loss in general, and to loss to ground and surface water in particular. Therefore, it is extremely important to be aware of which terms are included and their place within the balance, especially when comparing the balances of different farmers or farming systems. Moreover, surpluses may provide incorrect information concerning the scope for improving the operational management amongst a group of farmers when the population is heterogeneous in terms of imports and exports of nutrients across farm boundaries (Schröder et al., 2003).

Despite its shortcomings, the balance sheet approach has become a key element in the legislation of a number of European countries (Ambus et al., 2001; Happe et al., 2001; Neeteson et al., 2001). In these circumstances farmers are expected to balance their major inputs and outputs so that their annual surplus does not exceed a permitted threshold, which may be specified as a function of crop and soil type. Excesses may be fined with prohibitive fees or other types of penalties (Neeteson et al., 2001).

2.5. Nutrient surplus as a function of nutrient input

The nutrient surplus within different farm types is positively related to nutrient inputs (Jarvis, 1999; Watson et al., 2002) and thus summed nutrient inputs represent another indicator of losses to which threshold values have been linked in many countries (Heathwaite et al., 1993; De Clercq et al., 2001b). However, similar inputs may yield different surpluses across farm types and crop rotations. Even within one and the same farm type and crop type, similar inputs may result in different outputs, due to variation in husbandry techniques, crop characteristics, soil, climate and management (Schröder et al., 2003).

The assessment of inputs becomes even less indicative of losses if it focuses on the input of field-applied manure only. Such a focus ignores the variability of nutrient inputs through mineral fertilisers, atmospheric deposition and biological N-fixation among farms.

Nevertheless, the European Commission has selected the 'N input via manure' as an indicator and its threshold value has been set at 170 kg N ha^{-1} per year for nitrate sensitive areas (Anonymous, 1991). Most EU countries have already restricted the permitted application rate of manure, but discussions between the Commission and member states on how to designate nitrate sensitive areas still continue, as do the debates on corresponding threshold values. The Commission will only consider requests for deviation from the threshold value of 170 kg N ha^{-1} per year when based on objective criteria (length of growing season, predominance of crops with high N uptake, high precipitation surplus, high denitrification capacity). The present EU indicator has a relatively means-oriented character. Hence, whether it will result in sufficiently low nutrient losses without additional indicators and thresholds is uncertain. For example, the amount of P in many types of manure associated with 170 kg manure-N ha^{-1} per year, exceeds the annual P-uptake by most crops (Van Dijk and Sturm, 1993; Grubber and Steinwidder, 1996; Wright et al., 1998), which may eventually lead to considerable P-losses. It is uncertain whether this potential shortcoming of the present N-oriented EU threshold can be fully compensated for by gently formulated Codes of Good Agricultural Practice (e.g. Anonymous, 1991). Some European countries, therefore, have established an additional ceiling to the annual P-input through manure (De Clercq et al., 2001b; Neeteson et al., 2001; Steineck et al., 2001).

2.6. Nutrient input via manure as a function of livestock numbers

A simple look at databases of livestock density, nutrient surpluses and emissions across Europe confirms the negative relationship between the presence of livestock and environmental quality (Hofman, 1999; Hansen, 2001; Anonymous, 2001; Dame and Iestra, 2001; De Clercq et al., 2001b). The robustness of these relationships can be questioned, but it is not surprising that policy makers have focused their policies on regulating livestock densities as a first step.

As animal types strongly differ with respect to their nutrient excretion (e.g. Grubber and Steinwidder, 1996), threshold values for land spreading of either manure-N or -P have been transformed into standardised livestock densities per animal type. Subsequently, livestock densities have become the practical indicator on which control is based. Obviously, such a standardisation permanently fuels the debate on whether the transformation is correct: while farmers may argue that their adjusted dietary management resulted in lower excretion rates than the surmised standard value, environmentalists critically monitor live weight standards and excretion to see if selection pressure towards more productive animals has not taken place. The situation can become even more complicated when the transformation of excretion criteria to livestock density is based on N or on P only. When based on P, for instance, the adjustment of diets may indeed lead to lower P excretions without altering the N excretion. Such a focus on the emission of P might then lead to an increase in the permitted livestock density. This in turn may result in gradually increasing manure-N application rates, thus controlling P-losses at the expense of increased risks of N-loss.

Evidently, when calculating the application rate of manure from livestock density, corrections will have to be made for the manure exported from or imported into the farm. Even then, the relationship between the livestock density and the N input to soils is afflicted with variation due to differences in volatilisation of ammonia–N. Volatilisation is extremely variable due to differences in housing and manure storage, composition of diets, and the indoors and outdoors handling of manure (Bussink and Oenema, 1998; Monteny and Erisman, 1998; Paul et al., 1998; Kebreab et al., 2001). To our knowledge, the European Commission has not specified exactly how an application rate (including N excreted during grazing) of 170 kg N ha^{-1} per year should be converted into an acceptable excretion rate. Consequently, the interpretation currently varies between member states. German authorities, for instance, initially surmised that an effective application rate (i.e. *into* the soil) of 170 kg N ha^{-1} is equal to an excretion rate of 236 kg N ha^{-1} (Happe et al., 2001), whereas Dutch authorities reasoned that an effective application rate (i.e. *onto* the soil) of 170 kg N ha^{-1} is equal to an excretion rate of circa 200 kg N ha^{-1} (Oenema et al., 2000). Consequently, numbers of livestock allowed within one and the same animal type vary among EU members.

3. Discussion

3.1. Criteria for the selection of indicators

Regulation of nutrient management requires an indicator, just as speed is an indicator for traffic safety. The value of a particular indicator can be based on various criteria. Firstly, an indicator should be effective, i.e. it should be related to the intended objective: the indicator should be goal-oriented rather than means-oriented. Goal-oriented indicators leave entrepreneurs the greatest freedom to select only those measures they consider effective under their specific circumstances (Van der Werf and Petit, 2002). Secondly, it may be convenient that the indicator has an integral nature, i.e. it may be the sum total of other objectives, so that the total number of indicators (one for each separate objective) can be limited. Thirdly, an indicator should preferably be responsive and attributable to the (recent) actions of individuals. Only then will individuals almost instantly notice the impact of their (adjusted) management and the factors for which they can be held responsible. Finally, the indicator should be efficient, i.e. the costs of a sufficiently accurate measurement should be limited. The weighing of these criteria by policy makers can differ from that by farmers.

We conclude that no single indicator is likely to perform best for all of these four criteria simultaneously. Using regional groundwater quality as an indicator, for instance, accounts for numerous relevant sources of variation, but the indicator is hardly responsive to or attributable to the management of individual farmers, which makes it unattractive from a control point of view. When the regional water quality is too low, farmers will blame each other and other land users, while policy makers are not able to accurately specify whose nutrient management should be improved.

At the base of Fig. 1, livestock density is proposed as an indicator. The value of this means-oriented indicator can be accurately assessed on each farm at a low cost. Unlike the indicator above, which only addresses N and P concentrations in groundwater, (a reduced) livestock density implicitly encompasses various objectives simultaneously: (long-term) emissions of nitrate, nitrous oxides, ammonia and phosphorus, animal welfare (lower risks of disease dispersal, more room for grazing), promotion of biodiversity at field edges, and self-sustainability (lower demands for feed stuffs, agro-chemical inputs, and administrative, educational and financial support from outside). Thus, the indicator 'livestock density' has a more integral nature relative to the indicator 'N concentration in groundwater', for example. At the same time, livestock density has a relatively uncertain effect on the achievement of these goals and should be seen in terms of probability rather than of guarantee. Farmers can rightfully claim that a high livestock density, combined with good management, results in less N emission than a low livestock density combined with poor management (Schröder et al., 2003).

Generally, the effectiveness of the indicators presented in Fig. 1 decreases from top to bottom, whereas the integration of objectives as well as the responsiveness and attributability increase from top to bottom.

It is difficult to estimate the accuracy/costs ratio (i.e. the efficiency) of the various indicators. Indicators at the outer ends of Fig. 1 (except for the fluid term 'health and welfare') may be relatively inexpensive compared with indicators with large data requirements. For example, regular measurement of nutrient concentrations at a river mouth or in a drinking water well will be less costly than assessment of RSMN across the various spatial and temporal entities of an individual farm, or the input–output terms of nutrient balances. On the other hand, the latter kind of data is often readily available for book keeping purposes and for soil fertility management. The eventual costs of an indicator strongly depend on the required resolution and, when the demands are limited, will allow averaging and thus pooling of samples. Of course, there is a limit to the amount pooling that can be done, because it means that information about causes and desired actions will be lost. In addition, drinking water quality or ecosystem quality are not always served by apparent compliance when that has been achieved through averaging and dilution at a large spatial scale, just like traffic safety is not determined by the average speed of all participants together.

3.2. Harmonisation

A global harmonisation of environmental policy approaches to nutrient losses, including regulation, is to be generally welcomed. However, too much standardisation in the selection of indicators and threshold values may have scientifically questionable implications for which we once more refer to Fig. 1. When one really wants to take all the factors involved into account, the scientific evidence should be used to define differentiated target values for nutrient concentrations, surpluses, input levels or permitted livestock densities, just as speed limits imposed on traffic are not uniform. Consequently, some individual farmers and possibly entire countries may undoubtedly benefit from such a differentiation, whereas others may be confronted with serious local constraints imposed upon them by soil and climate. If we assume that such a differentiation is administratively feasible and affordable, it would imply that farming opportunities might not be equal for all. This can be seen as a serious objection, although the financial effect of reduced farming opportunities will eventually be reflected in the lower rent of land and facilities. In many debates on harmonisation reference is often made to the need for equal competitiveness. However, it is not self-evident why one should take this perceived need into account in matters of nutrient loss, when it is not done in matters of, for instance, water availability.

From a scientific point of view, harmonisation should primarily be directed at the development of a shared way of reasoning and standardised protocols, indicating exactly at which scale the achievement of objectives must be spatially and temporally evaluated. Harmonisation can therefore be served by the use of common indicators. This should not, however, be misinterpreted as a plea for standardised threshold values, because that would be scientifically questionable.

Just as there are scientifically sound reasons to differentiate threshold values within indicators, a regional differentiation of the indicators themselves may also need to be taken into consideration. In regions with a low production intensity, administratively simple indicators (e.g. livestock density) can suffice, whereas the use of complex, more expensive indicators like nutrient balances, may be an inevitable concession to the demands of evaluating environmental quality in a region opting for a high production intensity. However, this administratively disadvantageous position for intensive farms might not acknowledge that these farms can lose small amounts of nutrients per unit output despite their high nutrient loss per unit area. After all, small losses per unit output are a positive attribute with respect to global environmental effects, including land consumption (De Wit, 1992; Van der Werf and Petit, 2002; Neeteson et al., 2003; Schröder et al., 2003; Corré et al., 2003).

The previous section illustrates the ways in which harmonisation is quite an ambiguous concept. Harmonisation may pertain to a standardised treatment of either individual farmers or physical situations. The former maintains equal competitiveness, the latter a consistent and scrutinised back-casting from goals to means with discrimination between individuals for scientifically valid reasons.

4. Conclusion

Science is all about the acknowledgement of variation. Regulations based on sound science should therefore differentiate between different situations to do justice to their variation. However, society cannot wait until science has identified all the sources of variation and translated them into fair and detailed legislation, including variable indicators and threshold values. Moreover, there often have to be limits to differentiation, because the wish for sound science has to be balanced against the need for equal treatment of individuals and a reduced administrative burden. Politicians are faced with this dilemma. It is they who have the difficult task of weighing up the four criteria for indicator selection presented in this paper. They also have to adjudicate between the demands for a harmonised treatment of either individuals or physical situations. In addition to all this, they have to decide to what extent and at what spatial scale society must achieve a given level water quality or ecosystem quality.

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