Proceedings of International Conference on
**Realistic Expectations for Improving European Waters**

Final Conference of COST Action 869

# Mitigation Options for Nutrient Reduction in Surface and Ground Waters

12-14 October 2011, Keszthely
HUNGARY

Final Conference of COST Action 869

**Mitigation Options for Nutrient Reduction in Surface and Ground Waters**

# Proceedings of International Conference on **Realistic Expectations for Improving European Waters**

**Edited by**

Dr. István Sisák

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# Preface

The final conference of the COST Action 869 (Mitigation Options for Nutrient Reduction in Surface Water and Groundwaters) took place at Keszthely, Hungary between 12-14 October 2011 under the sobering title: **Realistic expectations for improving European waters**.

After the laborious process of manuscript writing and editing sixteen papers have been included in this proceedings which represents the broad geographic and thematic scope of the COST Action. Wim Chardon, the chair of the action provided a summary on the activities and pointed out eventual inconsistencies between research and regulation. The working group leaders and their co-authors gave overview on the progress in their specific area, exemplified major achievements and discussed questions which remained open for future research. István Ijjas presented background of the Strategy for the Danube Region. One paper called for drastic changes in the structure of animal husbandry in the EU but the next paper from New Zealand highlighted that such changes might imply unbearable costs. Two papers dealt with erosion in Slovakia and Hungary as the main mechanism of diffuse P transport, three papers investigated leaching and groundwater quality (UK, Germany and Bulgaria), two papers discussed river and oxbow water quality in Hungary and Romania and one paper studied effectiveness of mitigation options in Norway.

I would like to thank all authors, co-editors and supporters for their contribution and I would like to recommend this issue to the attention of the scientific audience.

Keszthely, 17$^{th}$ November 2013

István Sisák
Head of Organizing Committee, Editor

# COST ACTION 869 - MITIGATION OPTIONS FOR NUTRIENT REDUCTION IN SURFACE AND GROUND WATERS

*CHARDON, W.J.[1] – HAYGARTH, P.M.[2] – EKHOLM, P.[3] – SCHOUMANS, O.F.[1] – KRONVANG, B.[4]*

[1] ALTERRA, Wageningen UR, Wageningen, The Netherlands, e-mail: wim.chardon@wur.nl
[2] The Lancaster Environment Centre, Lancaster University, Bailrigg, Lancaster, UK
[3] Finnish Environment Institute, Helsinki, Finland
[4] Department of Bioscience, Aarhus University, Silkeborg, Denmark.

**Abstract**

During 2006-2011, COST Action 869, "Mitigation options for nutrient reduction in surface and ground water", was a research network financed by the EU Framework Programs. Support was given for organizing meetings and travel, but not for research. This paper gives an introduction of the Action, with the background and a summary of the main activities.

**Introduction**

The eutrophication of surface waters and the contamination of ground water due to elevated nutrient inputs have a serious impact on ecosystem health in many countries. Within the European Union, the Water Framework Directive (WFD) requires improvement of the quality of surface and ground water. This may require a drastic reduction in nutrient loss from agricultural land (KRONVANG *et al.*, 2005) with the related implications for the long term economic and environmental sustainability of agricultural systems. Since 1995, a series of International Phosphorus Workshops (IPW) was organized in Europe, every 3 years:

- IPW1, 1995 - Wexford, Ireland (TUNNEY *et al.,* 1997)
- IPW2, 1998 - Antrim, Northern Ireland (SHARPLEY *et al.,* 2000)
- IPW3, 2001 - Plymouth, England (HAYGARTH *et al.,* 2001)
- IPW4, 2004 - Wageningen, The Netherlands (Chardon and Koopmans, 2005)
- IPW5, 2007 - Silkeborg, Denmark (HECKRATH *et al.,* 2007; KRONVANG *et al.,* 2009)
- IPW6, 2010 - Seville, Spain (DELGADO *et al.,* 2010).

The next IPW meeting will be organized in 2013 (Sweden). The IPW meetings have greatly contributed to increasing our knowledge on the relations between agriculture and phosphorus (P) losses, and more specifically about the P transfer from soil to water, and the effects of mitigation measures. The international contribution of researchers to these workshops has led to a European network of researchers around the topic of P losses from agriculture. This network was strengthened within COST Action 832, "Quantifying the agricultural contribution to eutrophication", which ran from 1997 to 2003 (Chardon and Withers, 2003; Withers and Haygarth, 2007). Within COST 832, delegates from 18 countries were active (Austria, Belgium, Denmark, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Netherlands, Norway, Poland, Romania, Spain, Sweden, Switzerland, United Kingdom). The COST (European **Co**operation in the field of **S**cientific and **T**echnical Research) program is financed by the EU Framework Programs, and supports networks of researchers (see www.cost.esf.org). Support is given for organizing meetings, travelling, and dissemination, but not for research projects. The link between IPW and COST can be derived from the fact that during the first IPW meeting (1995) an announcement was made about the possibility to apply for a COST Action, by Paul Withers (UK), who became chair of COST 832 after the successful application. The first meeting of COST 832 was held parallel to IPW2 in Antrim (1998), and IPW6 (2010) was co-organized with Working Group 1 of COST 869. Moreover, the organizers of all IPW workshops were country representatives in COST 832 and / or COST 869.

**COST Action 869**

A follow-up of COST 832 started in November 2006: COST Action 869 "Mitigation options for nutrient reduction in surface water and ground waters" (see www.cost869.alterra.nl); the Action finished in November 2011. The main objective of COST 869 was to undertake a scientific evaluation of the suitability and cost-effectiveness of different options for reducing nutrient loss to surface and ground waters, at the river basin scale. This includes their limitations in terms of applicability under different climatic, ecological and geographical conditions. COST 869 focused on the steps that need to be taken within the EU Water Framework Directive in order to effectively reduce the nutrient losses from point and diffuse sources to surface waters and groundwater. The Action was undertaken in the context of balancing measures to reduce P losses with those necessary to reduce other nutrient losses such as nitrogen (N). Such measures are often in conflict, and need to be considered as part of an integrated program of measures.

At the end of the COST 869 the following 30 countries participated in the Action: Austria, Belgium, Bulgaria, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Israel, Italy, Latvia, Lithuania, Luxembourg, Netherlands, New Zealand, Norway, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, Turkey and the United Kingdom. The participating countries are shown in Figure 1, and the development of the number of countries during the course of the action is shown in Figure 2. The figure shows that at the start of COST 869 already more countries participated (21 + 3 intentions) than at the end of COST Action 832 (18). Thus, the research network created during COST 832 and four IPW-workshops made it possible for COST 869 to get going quickly.

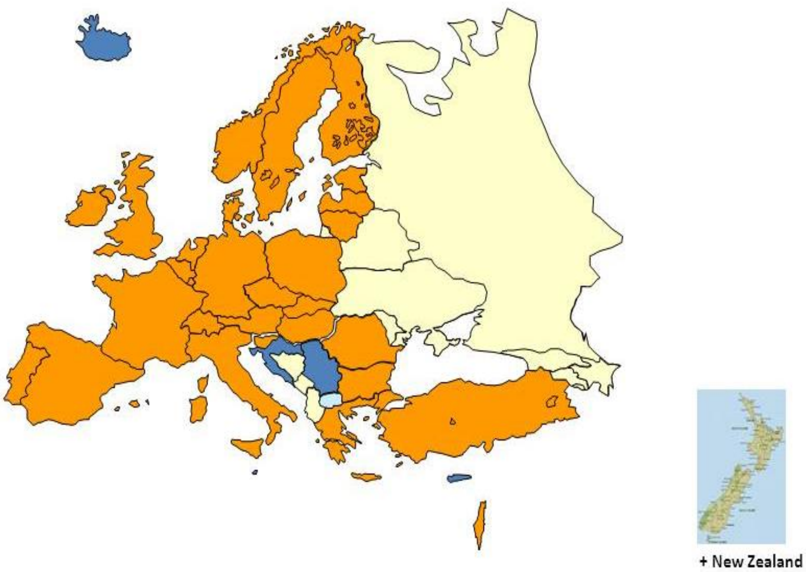



*Figure 1.* Countries participating in COST Action 869 (in brown)

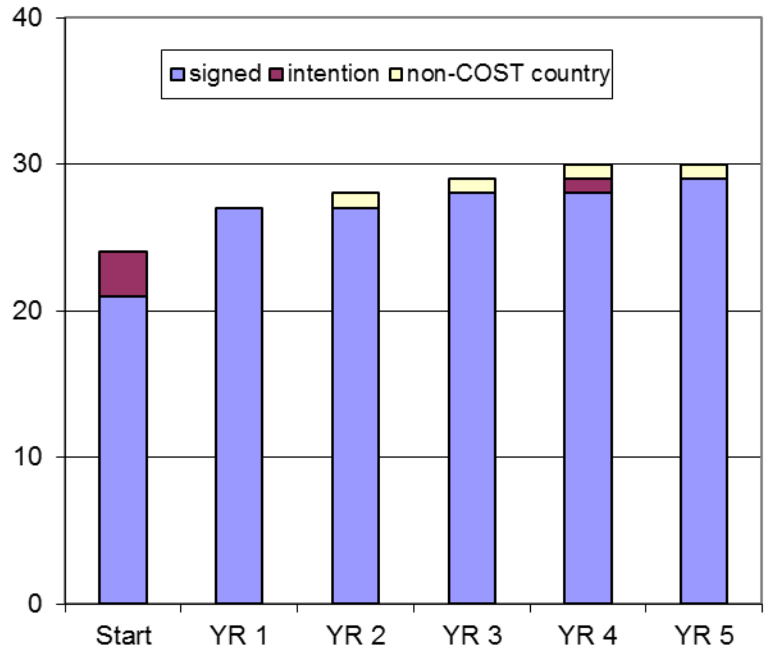



*Figure 2*. Development during the Action of the number of countries that joined COST 869 by signing the Memorandum of Understanding, or had expressed the intention to do this

**Management Committee (MC) and Working Groups (WG)**

Within COST 869, country representatives (max. 2 per country) were member of the Management Committee (MC). The MC was chaired by W.J. Chardon (Netherlands), and L. Heathwaite (UK) was vice-chair. The MC decided on how to spend the available budget, and on requests from new countries who want to participate in the Action. Most decisions were taken during 6 meetings: at the start and during every year (Brussels, Devon, Athens, Magdeburg, Seville and Keszthely). Between the meetings, 9 decisions were taken by the MC via a written (e-mail) procedure. Four working groups (WG) were active, their aims and the names of the chair and vice chair are given in Table 1.

*Table 1*. Working groups (WG) that were active in COST 869, their aims and the names of the chair and vice chair.

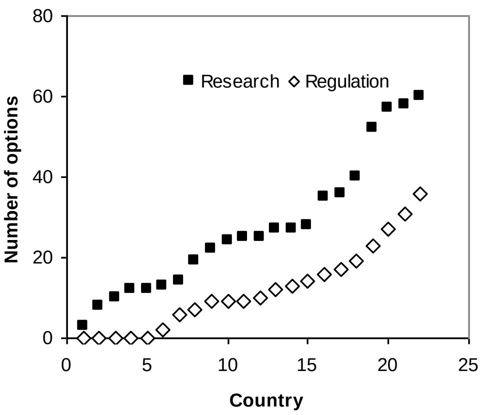
| WG | Aim | Chair | Vice-chair |
|---|---|---|---|
| 1 | Develop methodologies to localize critical source areas and transport routes in catchments | Heathwaite / Haygarth (UK)* | Litaor (IL) |
| 2 | Study the influence of nutrients on ecological processes in surface waters | Rekolainen / Ekholm (FI) ** | Skoulikidis (GR) |
| 3 | Evaluate different types of mitigation options | Schoumans (NL) | Lo Porto (IT) |
| 4 | Evaluate projects in example areas | Kronvang (DK) | Strauss (AT) |

* Heathwaite withdrew as chair of WG1 in 2009, Haygarth took over

** Rekolainen withdrew as chair of WG2 in 2007, Ekholm took over

**Inventory among participating countries**

In 2006, before the official start of COST 869, an inventory was made among representatives of countries that had shown interest in participating in the Action. A list of 88 mitigation options, taken from literature, was sent to representatives of these countries. They were asked if research had been done on these options in their country, and if the options were at that moment part of any regulation or of a recommendation. Reactions were received from 22 countries. In Figure 3 some results of the inventory are given, showing left for the 22 countries the large differences between the number of options on which either research has been done (varying from 3 to 60 options) or that are part of any regulation or of a recommendation (varying from 0 to 36 options). Figure 3 (right) shows that both numbers are correlated ($r^2$ = 0.58), and that on average about 50% of the researched mitigation options were implemented. However, in the lower region this relation is poor; some countries had done research on more than 20 options but did not have any option regulated at that time, and on the other hand one country that had 10 options regulated but where research on only 1 option was done.



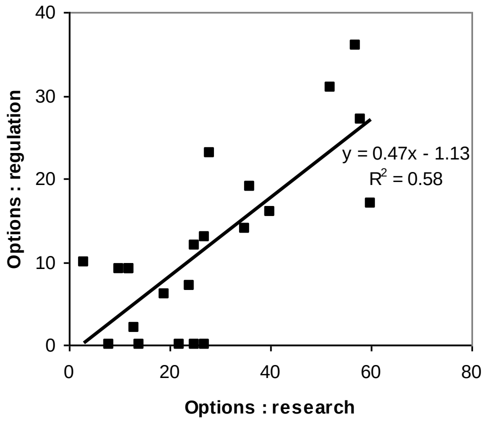




*Figure 3*. Results of inventory in 2006 among 22 countries participating in COST Action 869, showing (left) large differences between in number of options on which either research was done, or that were part of any regulation or of a recommendation. Right: the correlation between the number of options on which research had been done and that were part of any regulation

A conclusion drawn from the inventory was that at the start of COST 869 a large knowledge gap existed between the participating countries. Thus, knowledge transfers between countries were a main aim, e.g. via regular meetings organized during the Action, and via the Actions website (see www.cost869.alterra.nl/).

**Scientific meetings**

During the Action, 13 workshops were organized by the working groups. For these workshops, Table 2 gives the WG, the location, the number of participants, the countries represented, and the number of oral and poster presentations. In total there were 737 participations from, on average, 21 countries per meeting. There were 412 different participants from 40 countries. The COST program stimulates participation of female researchers and of persons in their early stage of their career (ESR; within 8 years after obtaining a PhD degree). For COST 869, 38% of the participants were female, and 43% were ESR. In total, 409 oral or poster presentations were given (61 % oral). For the scientific meetings one can find on the Action website (under 'Past meetings') a list of participants, all abstracts, and a pdf of most oral and poster presentations.

*Table 2.* Scientific meetings of four COST 869 working groups (WG), with number of participants, countries represented, and oral + poster presentations.

| WG | Location | Year | Participants | Countries | Oral | Poster |
|---|---|---|---|---|---|---|
| | | | | | | |
| 1 | Hamar, Norway | 2007 | 53 | 23 | 29 | 10 |
| 3 | Okehampton, Devon, UK | 2007 | 52 | 23 | 16 | 11 |
| 3 | Rome, Italy | 2008 | 24 | 17 | * | * |
| 4, 3 | Waidhofen, Austria | 2008 | 40 | 17 | 22 | 6 |
| 2 | Athens, Greece | 2008 | 41 | 20 | 16 | 4 |
| 2 | Keszthely, Hungary | 2009 | 38 | 18 | 24 | 6 |
| 3 | Wageningen, Netherlands | 2009 | 41 | 20 | 21 | 10 |
| 4 | Notwill, Switzerland | 2009 | 52 | 20 | 17 | 13 |
| 4 | Magdeburg, Germany ** | 2009 | 79 | 29 | 15 | 8 |
| 4 | Ballater, Scotland, UK | 2010 | 46 | 15 | 24 | 8 |
| 2, 3 | Jokioinen, Finland | 2010 | 51 | 18 | 23 | 19 |
| 1 | Seville, Spain *** | 2010 | 153 | 31 | 29 | 16 |
| 1,2,3,4 | Keszthely, Hungary | 2011 | 67 | 24 | 12 | 44 |
| | ***Total*** | | ***737*** | | ***248*** | ***161*** |
| | ***Average*** | | ***57*** | ***21*** | ***21*** | ***13*** |

* Factsheets on mitigation options were discussed
** joined WG1 meeting / COMLAND conference [only WG1 oral/posters mentioned]
*** joined WG1 meeting / IPW6 workshop [only WG1 oral/posters mentioned]

One of the scientific meetings (Rome, 2008), and 5 small meetings of WG3 were dedicated to developing a database with factsheets of mitigation options (see report by SCHOUMANS *et al.*, 2011). The process of this development is described in another paper in this series (SCHOUMANS *et al.*, 2012).

The visibility of the COST program, in particular of COST 869, was expanded by the meetings in Magdeburg, Germany (2009) and Seville, Spain (2010). The Magdeburg workshop was organized adjacent to the Conference on: "Land and Water Degradation - Processes and Management", organized by the commission of Land Degradation and Desertification (COMLAND), of the International Geographic Union (IGU). The meeting in Seville was organized together with IPW6 (see above). Persons from 12 countries not participating in COST 869 attended the Magdeburg meeting, for the meeting in Seville this was from 9 countries.

A series of 13 papers from a meeting of WG4 was published in the J. of Environ. Quality (issue 2, 2012), with an introduction paper by STUTTER *et al.* (2012). Publication costs, including the right for open access, were paid from the budget of COST 869. Another series of papers, presented in a WG1 workshop in Seville (2010), is to be published in Soil Use and Management. The special issue will also contain papers presented during the IPW6 workshop. Also these papers will have open access, and COST 869 has contributed to the publication costs.

Finally, a series of 5 papers, presented in Jokioinen in 2010, will be published with 3 related papers in the open access journal Agricultural and Food Science.

**Short-term scientific missions**

Within COST 869 budget was available for short-term scientific missions (STSMs), defined as a visit of (minimum) one week of a student or researcher from one country participating in COST 869, to a colleague in another participating country. The scope of an STSM can range from a visit to a partner country for exchange of ideas, preparatory discussion for larger research programmes, collation and analysis of data and a short periods of research. Funding was available for travelling and subsistence. Table 3 shows the visitors and hosts of the STSMs within COST 869. In total, there were 10 STSMs with visitors from 8 countries, during which hosts in 5 different countries were visited. The STSMs of C. van der Salm and C. Lévi to Denmark resulted in a joint publication by VAN DER SALM *et al.* (2011).

*Table 3.* List of short term scientific missions (STSMs) during COST 869

| **Visitor** | **Host** |
|---|---|
| T. Page, UK | M. Bechmann, Norway |
| A. Wagner, Germany | P. Groenendijk, Netherlands |
| C. van der Salm, Netherlands | G. Rubaek & B. Kronvang, Denmark |
| C. Hahn, Switzerland | T. Page, UK |
| C. Lévi, France | B. Kronvang, Denmark |
| Søren Larsen, Denmark | C. van der Salm, Netherlands |
| P. Kynkäänniemi, Sweden | C. Deasy, UK |
| K. Johannesson, Sweden | C. Deasy, UK |
| R. Uusitalo, Finland | E. Barberis and L. Celi, Italy |
| L. Beesly, UK | M. Marmiroli, Italy |

**Final meeting of COST 869 and future prospects**

During the final meeting in Keszthely Hungary, 2011, the chairs of the WGs presented an overview of the work done within the Action, and of the lessons learned in their WG. They also organized discussions on topics related to the WG, in four rounds. A summary of the lessons learned and of the outcomes of the discussions is given in separate papers in this issue (ZHANG, EKHOLM, SCHOUMANS, and KRONVANG *et al.*).

We may conclude that the research network around phosphorus and its losses to the aquatic environment, formed during COST 832 and the IPW meetings (see above), was expanded and strengthened in COST 869. We hope that a new COST initiative will be successful in the (near) future. Unfortunately, eutrophication of surface water has not been an item on the EC research agenda for many years. A statement was written by the MC of COST 869, and sent to the DG Research & Innovation, DG Environment, DG Agriculture and Rural Development and COST office. It emphasizes the urgent need of such a research agenda because it is foreseen that the objectives of the WFD will not be reached at the short term and for many countries also not in 2027. Furthermore, the expected global scarcity of P should receive more attention (SCHRÖDER *et al.*, 2011).

Preventing eutrophication by abating improper use of P will not only improve ecosystem health, but will also be beneficial for food security in the future. A new COST initiative could help to exploit the huge amount of knowledge available in the network created and maintained in COST 832 and 869, and during meetings in the IPW series.

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# LOCALISATION OF CRITICAL SOURCE AREAS IN CATCHMENTS ACHIEVEMENTS IN COST ACTION 869 WORKING GROUP 1

*Ting ZHANG*[1] *– Phil M. HAYGARTH*[1] *– M. Iggy LITAOR*[2] *– Wim J. CHARDON*[3] *– A. Louise HEATHWAITE*[1]

[1] Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK, tting.zhang@gmail.com
[2] Tel Hai College, Israel
[3] ALTERRA, Wageningen UR, Soil Science Centre, Netherlands

**Abstract**

The COST (Cooperation in Science and Technology) action 869: Mitigation options for nutrient reduction in surface water and ground waters consists of four working groups (WGs). This paper concentrates on the outputs and highlights from Working Group One (WG1), focused on localization of critical source areas in catchments and on the determination, quantification and verification of (hydrological) transport routes. It reviews the outcomes of all four workshops of WG1 including critical source area identifications, pathways of nutrients delivery, cost effective mitigation options, the choice of simple versus complex models, uncertainty and scaling problems, lag time issues, farmer participation and new ways of cross discipline working, networks and communications etc.

**Key words:** Water quality, diffuse pollution, phosphorus delivery, critical source area, mitigation options

**Introduction**

Legal requirements projected from Water Framework Directive (WFD) would pose new challenges for the assessment, protection and management of water quality of surface waters and groundwater in agro-catchments across Europe. Although sediments and attached nutrients are delivered from the whole catchment into rivers, not all locations in a catchment contribute equally even if they are in the same type of land use (HEATHWAITE *et al.*, 2000). One well cited finding is that ninety per cent of P transfer from agricultural land may be accounted for by 10% of the land area in 1% of the time (PIONKE *et al.*, 1999). This observation strongly suggests that fate and transport of P within an agro-catchment is governed by contributing zones defined as critical source areas (MAAS *et al.*, 1985; GBUREK & SHARPLEY, 1998).

In order to reach targets of WFD, a new COST (Cooperation in Science and Technology) action: Mitigation options for nutrient reduction in surface water and ground waters was launched in 2005 (http://www.cost869.alterra.nl/). It consisted of four working groups (WGs). This paper concentrates on the outputs and highlights from Working Group One (WG1), focused on localization of critical source areas in catchments and on the determination, quantification and verification of (hydrological) transport routes.

**Lessons from previous workshops**

Three workshops of WG1 have been held in the past few years in Hamar, Norway, 2007; Magdeburg, Germany, 2009; and Seville, Spain, 2010. During these workshops delegates discussed and reviewed existing methodologies used for identification of critical source areas of P loss in catchments. The pathways of P delivery within river basins have also been traced and assessed. Furthermore, different methods have been investigated at the locations in catchments where mitigation actions are most likely to be effective in terms of P loss reduction and cost. Some of the key messages and lessons learned follow.

***Methods for Critical Source Area (CSA) identification***

There are several methods for investigating the locations within a catchment that are most likely to be responsible for high rates of nutrients loss. Nutrient source, nutrient balance or index approaches are three ways that have been used to estimate nutrient pressure. However, most of the index approaches include expert/empirical elements or statistical modules and do not take CSA dynamics into account, hence more dynamic approaches are needed (TREPEL and OLLESCH, 2007).

COX *et al.* (2007) investigated different approaches from empirical to physically-based algorithms. As expected, models have been gradually increasing in complexity as the need to answer more detailed questions has arisen and where simplistic empirical rules were shown to be inadequate (Figure 1).

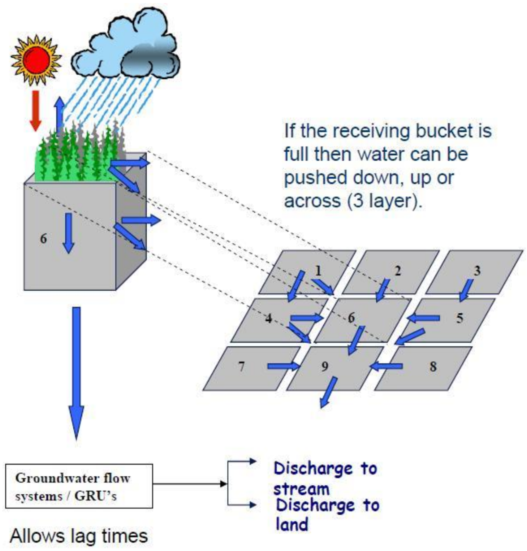


*Figure 1.* An example of a physically based distributed eco-hydrological model (COX *et al.*, 2007). The model operates at the grid cell level and includes sub-models such as energy fluxes linked to climate variability, crop growth, hydrological flow paths and spatial distribution of soil types and soil hydraulic properties

The SCIMAP project (REANEY *et al.* 2011) uses high-resolution land-use (farm holding) data and through inverse modelling against water quality monitoring data, provides an improved means for the identification, spatial prioritisation and assessment of the risks of farm/field scale sources of nutrient and sediment risk and their management at a catchment scale (Figure 2).

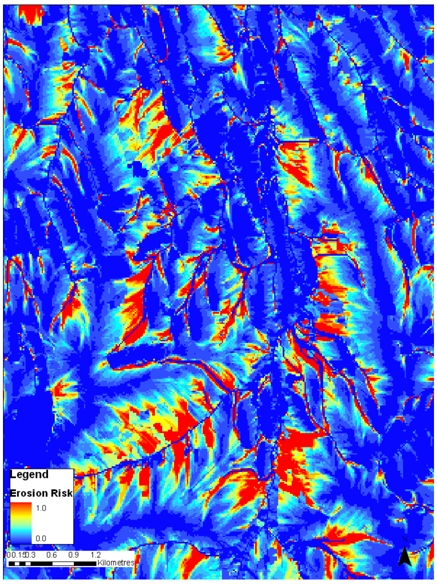


*Figure 2.* Erosion risk map by SCIMAP (picture from http://www.scimap.org.uk/scimap-fine-sediment/). Areas in red were identified as contributing fields because of their connectivity to streams and spatial pattern of land cover

***Pathways of nutrient delivery within river basins***

New technologies for tracing the movement of sediment across landscapes have been well developed in recent years. These new tracers allow us to tag soil particles using a chemical marker or to introduce particles to the soil that mimic its behaviour. Once applied to the soil the particles can be recovered from the landscape or fluvial system, and so the concentration of tagged particles present can be quantified. Therefore there is the potential to use different tracers or different 'species' of the same tracer to collect data on temporal and spatial patterns of soil redistribution on hillsides and sediment delivery to fluvial systems.

MICHAELIDES *et al.* (2010) and STEVENS and QUINTON (2008) addressed whether the potential of tag and trace technologies to unlock some of the mysteries of sediment transport could be fulfilled and presented a vision for the future development and application of these new additions to the erosion scientist's toolbox. Figure 3 shows the transit times and travel distances of different sediment tracers via a downslope in a lab erosion experiment after a storm event. It shows spatial patterns of all rare earth element (REE) oxides tracers at the end of the experiment obtained from the 0–5 cm cores. Filled circles denote final REE concentrations and open circles denote the initial REE concentration at the tagged plot.

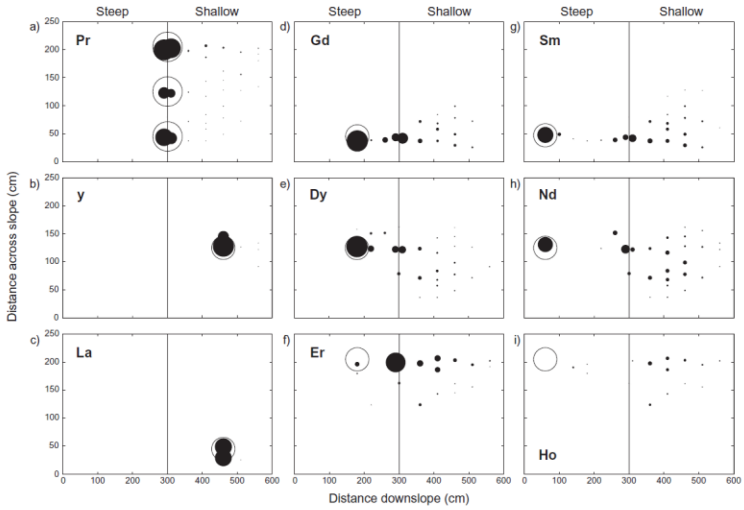


*Figure 3.* Spatial pattern of soil redistribution across a break in slope derived using REE tracers (taken from MICHAELIDES *et al.*, 2010)

***Locations of cost effective mitigation options***

Many researches have made contributions in finding out the cost effective mitigation options. HACIN (2007) investigated the variable effect of wetlands in modifying nutrient loadings, linking the P losses to the in-stream effect of uptake by macrophytes; this is an area which is seldom linked but has implications for cost effectiveness of mitigation measures as different water bodies have different sensitivities to P reductions. HAYGARTH *et al.* (2009) reported a range of mitigation options applied in England and Wales and provided a simple way to prioritize cost and effectiveness by plotting a cost curve (Figure 4).

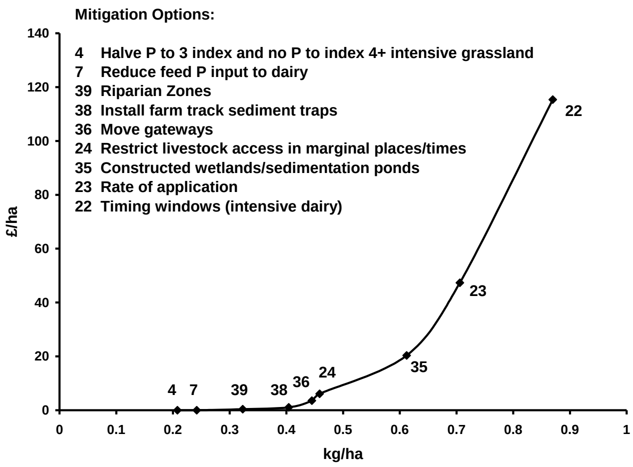


*Figure 4.* Example cost curve of mitigation options for intensive dairy farms in England (HAYGARTH *et al.*, 2009)

**New horizons raised by WG1**

The final workshop of WG1 was organized in Keszthely, Hungary on 12-14 October 2011. In this workshop, new horizons were explored followed by prioritization needs for the future research:

1) The choice of 'simple' versus 'complex' models;

2) Uncertainties and scaling problems;

3) Issues related to the lag time between intervention and catchment response;

4) Farmer participation in modelling;

5) New ways of cross disciplines working, communities, networks and communications.

***The choice of 'simple' versus 'complex' models***

It was determined that simple models are easy to use and better for a practical, focussed, application. However, complex models have advantages for furthering understanding and explaining the processes. The choice of a simple versus a complex model greatly depends on the aims and objectives of the project.

Models are used to simulate potential scenarios and to identify knowledge gaps in our perceptual concept or in our monitoring schemes. It has been suggested that model structuring should follow a procedure from a perceptual model of qualitative understanding to a conceptual model of equations to numerical solution using well designed algorithm (VADAS *et al.,* 2007). A flawed, inaccurate or incomplete perceptual model would result in an unreliable chain of prediction.

The choice of models depends on the problem needs to be solved and what kinds of factors have been put into consideration. It also depends on the spatial and temporal scales of the experiments. For example, the framework for diffuse pollution known as the transfer continuum (HAYGARTH *et al.,* 2005) and the popular P index computational methodology (HEATHWAITE *et al.,* 2003; BUCZKO and KUCHENBUCH, 2007) have been widely used as a perceptual model and a conceptual model respectively and showed reasonable success when applied in temperate agro-catchments (e.g., GBUREK *et al.,* 2000; BECHMANN *et al.,* 2005 among many others). However, when the P index algorithm was used in a Mediterranean catchment for extreme rain events, results showed the range of spatiotemporal variations must be included to account for the climate variability (LITAOR *et al.,* 2011). A key question in such undertaking is how complex the model structure should be to satisfy the needs in terms of management decisions and mitigations to be implemented. This issue is closely related to data requirements and monitoring schemes for a sound assessment of P spatiotemporal variations and model application.

***Uncertainties and scaling problems***

Uncertainty problems exist everywhere in diffuse pollution research. It can be found not only in data monitoring and sampling but also in understanding and modelling nutrient mobilisation and delivery processes. It is difficult to make an accurate determination of farm balance inputs because the animal manure recycling that goes on in agriculture. Furthermore, there is often not sufficient evidence to practically separate delivery from mobilization. Mobilization depends on spatial heterogeneities in soil properties, soil moisture conditions, vegetation cover, rainfall intensities and surface and subsurface flow rates. Delivery estimates is dependent on the estimates of mobilization. Therefore, errors in mobilization estimates will inevitably be transferred into delivery estimates. (BEVEN *et al.,* 2005)

Uncertainties are often connected to the scaling problem (HAYGARTH *et al.* 2012). The P mobilisation and delivery processes are much more complicated at the catchment scale than at the plot scale as the processes controlling the amount of P in surface water change and shift in their relative dominance (QUINTON *et al.,* 2003). It is reported that buffer strips at a catchment scale may change the dominant P form from total phosphorus (TP) to Filterable Reactive P (FRP) and that such changes may lead to limitations in the effectiveness of RBZs (MCKERGOW *et al.,* 2003). Without consideration of the uncertainty in our current ability to estimate the behaviour of diffuse pollutants, estimations of the P delivery, the effectiveness of mitigation measures and hence cost-effectiveness might be misleading (ZHANG *et al.,* 2012).

Phosphorus delivery estimation models are typically built using scientific knowledge gained at different spatial scales such as from laboratory experiments, plot and field scale studies. Differences across scales exist in hydrological processes, which drive P losses, leading to scaling issues when modelling diffuse pollutants (BEVEN *et al.,* 2005; DOUGHERTY *et al.,* 2004). Thus, results obtained at one scale may not be able to be directly transferred to another scale. As a result, modelling strategies must employ a scaling strategy, which in turn can introduce significant uncertainty.

***Issues related to the lag time between intervention and catchment response***

When land management strategies like mitigation options are applied in a catchment, there is a delay in time between the nutrient perturbation and the catchment response. In groundwater dominated catchments, this is caused partly by the delay in solute transfer from soil to watercourses. The lag time stops immediate detection of any water quality improvement (BURT *et al.,* 2008; BURT el al., 2011; HOWDEN *et al.,* 2009; JACKSON *et al.,* 2007; JACKSON *et al.,* 2008; KIRCHNER *et al.,* 2000; OWENS *et al.,* 1992). While in catchments dominated by more rapid surface and near surface runoff pathways, observed decreases in nitrate and P concentration are also much slower than have hoped for (BURT *et al.,* 2008; BURT *et al.,* 2011).

Lag time will vary according to different mitigation measures. Delivery control methods, like wetlands, will see the P delivery reduction being reasonably rapid, but source control methods like buffer strips may take several years to see the effect. In this case, the financial cost is essential to consider when installing mitigation measures. Due to the lag issue, the long term effect of mitigation measures, especially on P delivery, may take several years to see the difference before and after. This makes it difficult to convince stakeholders such as farmers about the effectiveness of mitigation measures, when the effects of 'cause' and 'response' are so difficult to link.

***Farmer participation in modelling***

Farmers are the key land owners who are able to understand and potentially manage the diffuse pollution in the landscape. Thus their participation in the Critical Source Areas research is invaluable, as they can provide insightful information about their local land and potential farming practice situations.

In the UK the Government Defra has established a national network of demonstration test catchments. In one example, the Eden Demonstration Catchment (EdenDTC) use farmer networks and one example is a meeting in June, 2011. The farmers showed an interest in the data being collected and were keen to talk about the interpretation of the information, what it might mean for them and even offered to host monitoring equipment on their land. Many farmers were keen to participate in the project and a number of them have offered to keep diaries on their management decisions in order to capture useful additional information to supplement the data gathered at the monitoring stations and improve its interpretation. (From Eden DTC website, http://www.edendtc.org.uk/2011/10/meeting-with-farmers-in-morland/)

***New ways of cross disciplines working, communities, networks and communications***

The WG1 delegates are keen to link all the models of WG1 and WG3 across the WG barriers and boundary. This is because WG3 focuses on mitigation options, which is closely related to WG1. Patrick Wallman from Sweden introduced the Hydrological Predictions for the Environment (HYPE) open source community on 23rd Nov 2011, Stockholm. This can elaborate on how to best design an open source community in hydrology based on international experiences in hydrology and related disciplines. The HYPE model is officially used by the authorities in Stockholm and its source code is open to all users in hydrology, hydrological modelling, hydrological data distribution and source code development – e.g. scientists, authorities and consultancies. Similarities exist with the UK NERC Environmental Virtual Observatory approach (www.evo-uk.org).

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# INFLUENCE OF NUTRIENTS ON ECOLOGICAL PROCESSES IN SURFACE WATERS ACHIEVEMENTS IN COST ACTION 869 WORKING GROUP 2

*Petri EKHOLM*[1] – *Vera ISTVÁNOVICS*[2]

[1] Finnish Environment Institute, P. O. Box 140, FI-00251 Helsinki, Finland, e-mail: petri.ekholm@environment.fi

[2] Water Research Group, Budapest University of Technology and Economics, Department of Sanitary Engineering, H-1111 Budapest, Hungary

**Abstract**

We discuss the link between ecological processes and solutions designed to maintain or achieve good ecological state in European waters. Special emphasis is placed on three themes: (1) what are the key factors controlling primary productivity in various types of surface waters, (2) can we speed up the achievement of the target state by system manipulations using well-established or novel restoration techniques, and (3) how should we define, evaluate and monitor the ecological quality of waters in the context of the River Basin Management Plans that translate the objectives of the Water Framework Directive into management measures? The paper is based on the discussions held in the Working Group 2 of the COST Action 869.

**Keywords:** Lakes, Rivers, Management, Water Framework Directive, River Basin Management Plan

**Introduction**

The aim of Working Group 2 of the Cost Action 869 was to examine the link between ecological processes and solutions designed to maintain or achieve good ecological state in European waters, as demanded by the Water Framework Directive (WFD). At four meetings, the following focal questions were discussed:

1. What are the key factors controlling primary productivity in various types of surface waters?
2. Can we speed up the achievement of the target state by system manipulations using well-established or novel restoration techniques?
3. How should we define, evaluate and monitor the ecological quality of waters in the context of the River Basin Management Plans that translate the objectives of the WFD into management measures?

Below, we give a brief outline of the conclusions of these discussions and evaluate the success of the Working Group in enhancing scientific networking. Note that this article is a subjective interpretation, and cannot be taken as a statement of the COST Action.

**Results and discussion**

***Which factors control primary productivity in surface waters?***

Nutrient limitation is used with two different meanings often without making a proper distinction (REYNOLDS, 1992; ISTVÁNOVICS, 2008). *Growth limitation* refers to the factor that constrains the instantaneous growth rates of primary producers thereby acting as one of the key selective factors among species with different adaptive traits. By contrast, in the case of *biomass limitation,* the composite biomass of a community, such as the phytoplankton, cannot increase due to the shortage of a resource, i.e., the system has exhausted the supportive capacity of the environment. Eutrophication management aims at enhancing biomass limitation.

Phosphorus and nitrogen are the two nutrients that most often, but not always, determine the supportive capacity of surface waters (VOLLENWEIDER and KEREKES, 1982). Therefore by reducing the availability of these nutrients, primary production can be lowered in waters where the supportive capacity was nutrient determined in their pristine state. The response of the system, however, is inevitably delayed due to a regime shift. The length of the delay is determined by the ecological resilience (food web structure, presence of cyanobacteria) and enhanced benthic release of nutrients (a transient disequilibrium between diminished external loads and nutrient-rich sediments formed under higher external loads; SAS, 1989). Besides phosphorus and nitrogen, other nutrients may limit the biomass of primary producers in certain types of waters and for certain types of biota. As an example, carbon can be exhausted in extremely soft waters and is central for heterotrophic humic systems, the biomass of diatoms may often be silicon-determined and iron is suspected to limit productivity in oligotrophic oceans. Moreover, a group of compounds acting as electron acceptors (nitrate, sulphate, the oxides of iron and manganese) greatly modify the cycling of nitrogen and phosphorus. For example, the tendency of nitrogen to be the limiting nutrient in coastal waters can be addressed to the effect of sulphate on the phosphorus cycling (BLOMQVIST *et al.,* 2004, LEHTORANTA *et al.,* 2009).

Although a wide range of physiology-based methods have been developed and tested to identify the limiting nutrient, these labour-intensive procedures do not suit routine monitoring. Therefore, environmental concentration of nutrients is commonly used to assess potential biomass limitation for managerial purposes. This approach faces many difficulties including the fact that nutrients are present in various forms, the chemical identification of which may be challenging and the availability of which is uncertain. We do not have universal rules to determine, whether the concentrations and loads of dissolved or total nutrients provide more relevant information for the management. While it is usually more straightforward to consider total nutrient loads at an early stage of management, when the share of sewage loads is overwhelming, a shift to more uncertain and more disputable "biologically available" nutrient loads may be useful at a later stage, when diffuse loads become predominant. Additionally, total phosphorous is generally thought to be a proxy of human pressure and especially currently on-going intercalibration procedure uses mostly total phosphorus as an estimate of human impact (POIKANE *et al.,* 2010).

One should keep in mind that nutrient ratios are meaningless when the *absolute* concentration of the relatively less abundant nutrient is so high that its limiting role can be excluded. REYNOLDS (1992) argues that growth limitation of lacustrine phytoplankton can be excluded, if the concentration of dissolved reactive phosphorus exceeds 5 mg $m^{-3}$, and that of nitrate exceeds 200 mg $m^{-3}$. Obviously, there are systems in which primary production is controlled or modified by other factors than nutrients: e.g., humic substances and total suspended solids affect both light availability and the spectral properties of light in the water. Often this natural phenomenon is neglected and stained waters are classified to worse ecological condition than expected. The topic of nutrient limitation is dealt in more detail in a factsheet downloadable on the website of the action (http://www.cost869.alterra.nl/FS/FS_NPratio.pdf).

***Ecological responses to system manipulation***

Several techniques have been developed to manipulate ecological processes in surface waters. Three broad categories of methods can be identified: (i) physical interventions, like modification of the hydrological conditions, dredging, destratification (ii) chemical methods, like treatment with by Al, Fe or Ca compounds, and (iii) biotic methods, like food web manipulations by removal of fish or introduction of predatory fish. Regardless of the technique, reduction in external nutrient loads should precede internal manipulation. The restoration measures should be sufficiently intense, continuous and simultaneously flexible enough to integrate the feedback on the system response. A well designed, detailed and sustained monitoring is unavoidable to provide this feedback. Uncertainties and the time scale of expected recovery should be shared with stakeholders. Several review papers discuss the outcome of system manipulations (e.g. SØNDERGAARD *et al.,* 2007, 2008).

***Nutrient standards and ecological indicators used in the River Basin Management Plans***

The WFD put aquatic ecology at the base of management decisions, and thus, River Basin Management Plans must make decisions based on the response of aquatic organisms to environmental stress (HERING *et al.,* 2010). Yet, ecosystem processes are sometimes weakly integrated into the River Basin Management Plans (Fig. 1 shows an example of the discrepancy between variables used to quantify the pressures and the effects in standing waters). Reasons for this weak integration are manifold and may include the difference in the educational background and perception of ecologists and administrators, enormous complexity of ecosystems, gaps in our understanding of the non-linear and delayed relationships between stressors and system responses. Observations on physical, chemical and biotic elements of water quality are restricted in space and time, posing the challenge of up-scaling the available information to increasingly higher levels of hierarchy from individual water bodies, to river networks, catchments and transboundary basins, each having their characteristic spatial and temporal scales. To make the task even more complicated, we should design, implement and evaluate the efficiency of management actions in a changing climate, where the frequency of extreme events (e.g. storms, floods, droughts) increase and shifts in timing of processes occur both in terrestrial and aquatic ecosystems. There are indications that ecosystems respond more sensitively to extreme values of key variables than to their mean values (HARRIS, 1999).

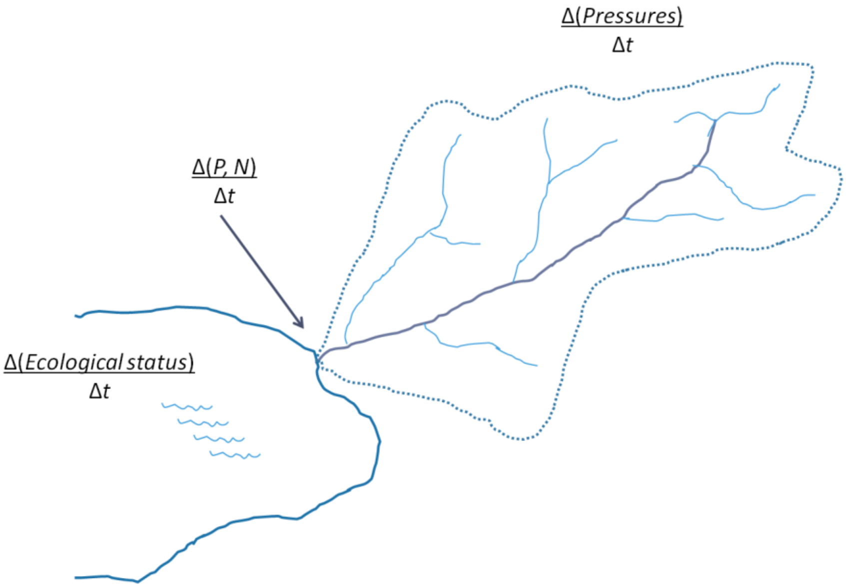


*Figure 1.* Water Framework Directive emphasizes ecological status, whereas monitoring of pressures is still largely focused on chemistry. Do we have tools to assess the pressures-impacts continuum?

There appears to be a lot of variation across Europe as to how the indicators of ecological state of water bodies are used. In some countries, classification is still primarily based on supportive variables (total nitrogen, total phosphorus) with less emphasis on the biotic quality elements than required by the WFD. Nutrient concentrations are often powerful indicators, and cheaper to analyze than to sample and identify species composition of various communities. However, biological indicators are often more sensitive indicators of long-term changes at an ecosystem level (POIKANE *et al.,* 2011). Noticeably, even relatively prosperous countries may lack the appropriate number of highly qualified taxonomists who are able to fully undertake the task required by the WFD.

Concerning nutrients and other chemical components, there is an uncertainty, which statistics are the most relevant in terms of ecological response (e.g., mean, minimum, maximum)? Yet, environmental degradation is often due to other factors than nutrient loads (e.g. contamination, serious hydromorphological modification, cutting down connectivity between habitats, etc.). Moreover, biological variables may react at an early stage of degradation, when chemical variables may not yet show any change. Interestingly, there is little or no effort to integrate well established, classical indicators such as daily oxygen curves into the arsenal of WFD compatible methods. Daily oxygen curves reflect the joint metabolic activity of aquatic communities, and recent developments in self-contained, automated, remote sensors have made it easy and relatively cheap to continuously measure concentrations of dissolved oxygen at time intervals of less than 1 min for periods from several days to much longer (STAEHR *et al.,* 2010).

Independent of the indicators selected, sampling frequency and duration should allow statistically sound trend tracking. Considering complexity of aquatic ecosystems and recognizing that different indicators vary in both their sensitivity and uncertainty, it seems risky to identify system state with the worst indication, as is done according to the One-Out All-Out principle. Some member states like Finland have extended the principle to cover individual biological quality element only to cases when available data is representative and individual indicator represents the entire biological community (e.g. RASK *et al.,* 2011).

Systems vary in their response to load reduction and other measures. Therefore, indicators should reflect the outcome of management actions. However, in some cases the indicators do not reflect even drastic changes in pressures. We can neither exclude the opposite case. There appears to be gaps in our knowledge of the functioning and behaviour of aquatic systems under different pressures and exhibiting an array of natural characteristics. Adaptive management aided by proper monitoring offers a possibility to adjust River Basin Management Plans (updated in every 6th year till 2027).

During the implementation of WFD, differentiation between anthropogenic pressures and trends in natural variables is a crucial question. Models are helpful in source apportionment, provided that sufficient long-term monitoring data are available to feed the models with inputs. The higher the stochastic variability either in the drivers (e.g. climate) or in the examined system (e.g. deep *versus* shallow lakes, lakes *versus* running waters), the longer and more frequent data series are needed to test, whether target loads and the target state of water bodies have been met.

The current economic crisis in Europe may have two-fold implications. On the one hand, it may be beneficial for water quality, if it decreases excessive fertilizer use and supports sustainable use of manure. Central East European countries will not enjoy this benefit, since both fertilizer and manure applications rates went drastically down during their lasting economic crisis and the phosphorus balance of soils turned negative during the last decade (CSATHÓ *et al.,* 2007). On the other hand, economic depression may delay the implementation of costly management measures. Finally, unintentional changes have often had a larger effect on aquatic systems then measures performed on purpose (*cf*. the water quality effect of the lasting economic crisis; CSATHÓ *et al.,* 2007). As an example, future demand for food and biodiesel may easily become an unwarranted interference with the implementation of WFD.

In conclusion, by focusing on biological indicators, WFD may widen our perspective on the degradation of aquatic systems. Especially hydrologically and morphologically heavily modified waters, like hydropower reservoirs, might support pristine water quality, whereas their biological communities are degraded. It appears that the implementation of WFD is interpreted differently among the EU countries. There may be a conflict among different EU policies that imply different land use concepts, e.g., WFD and the Common Agricultural Policy. Finally, WFD gives the same weight to all natural water bodies except for those, which are classified as "heavily modified" for human use like irrigation, hydropower or water borne traffic. Some countries have taken the risk of paying special attention to their ecologically most valuable waters that support high diversity and many endemic species. Should we follow the contemporary approach of nature conservation that has already made its evolution from protecting species to protecting habitats and presently to selecting the optimal combination of habitats based on their availability, quality and connectivity? Analogous approaches have recently been suggested for setting assemblage level conservation priorities in riverine networks (ERŐS *et al.*, 2010).

**Conclusions**

Water-related problems in Europe are manifold, ranging from nutrient enrichment and related ecological disturbances to degradation of habitats due to different types of human pressures like construction. With limited resources, we should find the most cost-efficient solutions for these problems. Prevention of environmental degradation needs permanent precaution but is less costly than the management of degraded ecosystems. Moreover, nutrients are valuable natural resources, which should not to be wasted as easily as today (TURTOLA *et al.,* 2010). Sound understanding of the structure and functioning of the ecosystem in question is a key prerequisite for selecting the cost-efficient strategies for appropriate catchment management. Collaboration of scientists with different backgrounds is highly encouraged, since our partial expertise must be synthesized to match the wholeness of nature (i.e. interacting ecosystems). The COST Action promoted scientific discussions on the topic and helped to understand the differences in both the present conditions and prevailing approaches across European countries. Furthermore, the Action efficiently catalyzed research networking and made possible for young scientists and scientists from Eastern Europe to present their work.

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# OPPORTUNITIES FOR MITIGATION OPTIONS TO REDUCE NUTRIENT LOSSES FROM LAND TO SURFACE WATER
# ACHIEVEMENTS IN COST ACTION 869 WORKING GRPUP 3

*SCHOUMANS, O.F.* [1,]* – *LO PORTO, A.* [2] – *CHARDON, W.J.* [1]

[1] ALTERRA, Wageningen UR, Wageningen, The Netherlands *oscar.schoumans@wur.nl
[2] Water Research Institute IRSA-CNR, Via De Blasio 5, 70132 Bari, Italy

**Abstract**

Within COST Action 869, called "Mitigation options for nutrient reduction in surface water and groundwaters", Working Group 3 aimed to collect, describe and evaluate mitigation options. During a series of meetings a setup for a description of options in factsheets was discussed, the factsheets were written and made available, a background report was published and a web-based tool was developed. This paper describes this process and its background, the setup of the factsheets, and the way they are grouped in different categories.

**Introduction**

The role of nutrients phosphorus (P) and nitrogen (N) in the eutrophication of surface water has already recognized in the mid twentieth century (REDFIELD, 1958). Negative effects of eutrophication caused a reduced functioning and biodiversity of aquatic ecosystems and surface water quality. Toxic substances produced by blue-green algae may cause fish kills, animal and human diseases (BURKHOLDER, 1998; SMITH *et al*., 1999). Avoiding these potential harmful effects, and reducing nutrient losses to the aquatic environment has received much attention. Examples are e.g. the series of International Phosphorus Workshops (IPW, see CHARDON and SCHOUMANS 2012, this issue) and the SERA-17 network of research scientists, policy makers, extension personnel, and educators that develop and promote innovative solutions to minimize phosphorus losses from agriculture in the USA (http://www.sera17.ext.vt.edu/). Due to improved wastewater treatment, phosphorus losses from point sources has decreased, and reducing loss from diffuse sources now receives more attention (SHARPLEY and WITHERS, 1994; SHARPLEY *et al*., 1994, 2001; Withers and Jarvis, 1998). Within the European Union, this process is accelerated by the Water Framework Directive (WFD) that requires improvement of the quality of surface and groundwaters (KRONVANG *et al*., 2005).

Within COST Action 869, Working Group 3 aimed to collect, describe and evaluate mitigation options. This paper describes the process of developing a database with such options during subsequent meetings of WG3. Thereafter, the setup of the factsheets, the division in categories and a tool for selecting the most relevant factsheets are presented.

**Development of the database**

The first WG3 workshop was organized in North Wyke, Okehampton, Devon, UK (November 2007). The topic of the meeting was: *Mitigation options: framework, effectiveness, and interactions*. The meeting was attended by representatives from 22 COST-countries, and during presentations a broad overview was given on measures taken in Europe for reducing N and P losses. During the meeting, results were presented from an inventory made in 2006, before the official start of COST 869, among representatives of countries that had shown interest in participating in the Action. A list of 88 mitigation options, taken from literature, was sent to representatives of these countries. They were asked if research had been done on these options in their country, and if the options were at that moment part of any regulation or of a recommendation. Reactions were received from 22 countries (CHARDON and SCHOUMANS, this issue).

Table 1 shows the options mentioned most frequently by the responding countries based on research being done. For these options it was also shown in how many countries the option was part of any regulation, which is much less than the number for research. Thus, for these options the research had not yet led to regulation in most countries. Table 2 shows the options mentioned most frequently based on regulation, and also the number of countries in which research on that option was done. In this case, the numbers are more or less equal, so for these options regulation is based on research in the same country.

*Table 1*. Mitigation options most frequently researched, according to an inventory (2006) among 22 countries, with the number of countries in which the option was regulated

| Option | Research | Regulation |
|---|---|---|
| Minimum, ridge or no tillage | 18 | 4 |
| Reduce fertilizer inputs to arable land | 16 | 7 |
| Reduce fertilizer inputs to grassland | 15 | 6 |
| Improve soil organic matter content | 14 | 3 |

*Table 2.* Mitigation options most frequently regulated, according to an inventory (2006) among 22 countries, with the number of countries in which the option was researched

| Option | Regulation | Research |
|---|---|---|
| Cover cropping during winter | 14 | 14 |
| Timing windows for manure application | 13 | 10 |
| Reduce rate of manure application and redistribution | 12 | 12 |
| Incorporation of manure | 11 | 11 |

The inventory also showed a large gap in knowledge on mitigation options between countries participating in COST 869 (CHARDON and SCHOUMANS, this issue), and filling this gap was considered as a major target of the Action. Therefore, during a discussion it was decided to develop a database with descriptions of mitigation options, called factsheets, written by participants in the Action and to be published on the website of the Action. It was felt that summarizing information in factsheets, and putting this together in a European framework, would have some major advantages: (i) reduce duplication of work; (ii) data mining and sharing information and methodologies; (iii) transferable information within EU; (iv) obtaining evidence about effectiveness in relation to geographical conditions and scale issues.

During a small meeting in Amsterdam (February 2008) a setup of the factsheets was discussed, a division of mitigation options in 8 categories (with subcategories) was defined, and a full meeting in Rome was prepared.

Examples of descriptions of mitigation options served as the basis of the discussion, and were found in a report from the UK with 44 options (CUTTLE *et al*., 2007; CHERRY *et al*., 2008), a report with 22 options (in Danish, SCHOU *et al*., 2007), and descriptions of 32 Best Management Practices, produced within the SERA17 framework mentioned above. After the meeting, participants in the Action were asked to write draft factsheets, or to coordinate the writing of factsheets on a certain category of mitigation options.

Before the full meeting in Rome (April 2007), 60+ draft factsheets were prepared. During the meeting, decisions were taken on the setup of the factsheets and the division in categories. It was also decided that (i) the factsheets will be written for a public with a higher education, like water board managers, policy makers, advisers, and researchers / the participants of the COST action themselves, and (ii) that a general introduction to the list of factsheets has to be written, describing the background of the fact sheets and the different categories. A discussion was started about a conceptual framework the mitigation options fit in, based on a model concept of the source - mobilization - transport - impact continuum defined by WITHERS and HAYGARTH (2007).

During a meeting of WG4 in Waidhofen/Ybbs, Austria (May 2008), and of WG2 in Athens (September 2008), the results of the Rome meeting were presented, and attendees were asked to volunteer in reviewing factsheets. During small meetings in Wexford (November 2008), Ghent (December 2009), and Amsterdam (March and October 2010), the work on factsheets and on a background report was further discussed and completed.

**Setup of factsheets**

Each factsheet contains the following paragraphs with information about the mitigation option:

- *Description, including if effect is aimed at nitrogen (N) or phosphorus (P)*
- *Rationale, mechanism of action;* the mechanism via which the option is able to retain N (e.g. via denitrification) or P (e.g. via sorption).
- *Relevance, applicability & potential for targeting;* the type of sources the option can be applied, e.g. on poultry farms, or within horticulture.
- *Effectiveness, including uncertainty;* an estimation is given how effective the option can be, under which conditions it will be most effective and under which this will be least.
- *Time frame;* this indicates if the option is assumed to be effective on short, medium or long term.
- *Environmental side-effects / pollution swapping;* some options may lead to unwanted effects in other environmental compartments, e.g. reducing nitrate leaching via denitrification may lead to release of $N_2O$ (greenhouse gas) or of P.
- *Administrative handling, control;* this describes if it is complicated to implement an option, and if application can easily be controlled.
- *Costs;* no estimation of real costs is given here, since costs can vary strongly between countries or even parts of a country. For example, labor costs differ strongly, and the price of land is highly dependent onpopulation density in a region.
- *References;* whenever possible references are given that are easily accessible.

**Setup of database - categories**

Different management strategies can be distinguished to reduce the nutrient losses and to improve the water quality. Therefore, the factsheets were grouped into eight management categories and linked to the different systems which influence the nutrient losses to surface waters and the impact on surface water quality (SCHOUMANS *et al*., 2011; Figure 1). The eight reported management categories are given in Table 3, with the number of factsheets. Note that since some factsheets will be combined in the (near) future the numbers in table 3 are indicative.

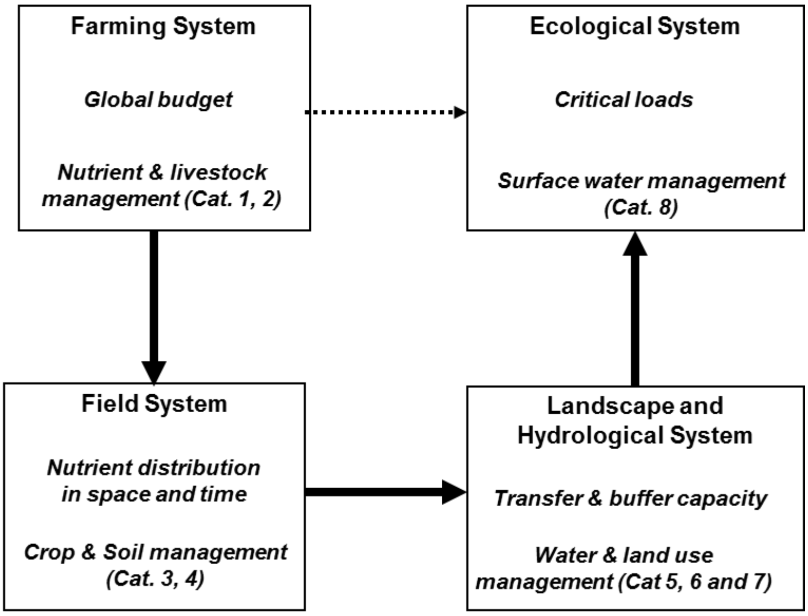


*Figure 1.*

*Table 3.* Main categories of mitigation options used in database, with number of factsheets in category

| | Main category | n |
|---|---|---|
| 1 | Nutrient application management | 25 |
| 2 | Crop management | 1 |
| 3 | Livestock management + production of minerals in manure | 7 |
| 4 | Soil management | 18 |
| 5 | Agricultural water management | 11 |
| 6 | Land use change | 1 |
| 7 | Land infrastructure | 8 |
| 8 | Measures in surface water | 9 |

Categories 1 to 4 are most related to agricultural practices that can be influenced by the farmer. Category 5 depends on possibilities and necessity to drain or irrigate land. Category 8, Measures in surface water, was added since, for improving ecological quality of surface water, it can be necessary to take (temporal) additional measures, even when export of nutrients from agriculture is drastically reduced.

**Background report**

A report was written that gives the background of the mitigation measures described in factsheets (SCHOUMANS *et al*., 2011). In total, 34 persons from 11 countries contributed to one or more chapters in the report. The report contains chapters with (i) a background of COST 869 and of the factsheets; (ii) a conceptual framework of nutrient losses; (iii) a summary of environmental European legislation in relation to agriculture, and (iv) a text on each category of options mentioned in Table 3.

**Access to factsheets**

All current factsheets were written in English, by 27 persons from 14 countries, and are available via the website of the Action: www.cost869.alterra.nl under "List of options and factsheets". There is no restriction on access to the database, so it is not limited to participants of COST 869. Via this website also a link is given to the SERA17 Best Management Practices mentioned before.

During the final meeting of COST 869 in Hungary (October 2011) it was discussed how end-users could most easily make a selection from the large number (70+) of factsheets that were published. It was decided to develop a web-based tool, and this idea was further elaborated during a small meeting in Amsterdam (December 2011). Work on this tool is in progress, and it will be made available via the website mentioned above.

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# EVALUATION OF PROJECTS IN EXAMPLE AREAS ACROSS THE EU ACHIEVEMENTS IN COST ACTION 869 WORKING GROUP 4

*Brian KRONVANG[1] – Peter STRAUSS[2] – Marc STUTTER[3] – Søren LARSEN[1] – Christian STAMM[4]*

[1*] Department of Bioscience, Aarhus University, Silkeborg, Denmark.

[2] Bundesamt für Wasserwirtschaft, Inst. Kulturtechnik und Bodenwasserhaushalt, Petzenkirchen, Austria.

[3] The James Hutton Institute, Aberdeen, AB15 8QH, Scotland UK

[4] Dept. Water and Agriculture, Swiss Fed. Inst. for Environ. Sci. and Technol., Duebendorf, Switzerland.

* Corresponding author address: Brian Kronvang, Department of Bioscience, Aarhus University, Vejlsøvej 25, DK-8600 Silkeborg, Denmark. E-mail: bkr@dmu.dk, Phone: +45 20833413.

**Abstract**

The main aim of COST Action 869 Working Group 4 was to support and strengthen a European-wide understanding between scientists and catchment managers on how nutrient losses from diffuse sources can be successfully combated by implementing different mitigation measures in River Basin Management Plans. This analysis was achieved during three workshops focusing on topics like: i) Evaluation of Projects in Example Areas across Europe; ii) Evaluation of projects in example areas: The Swiss Midland Lakes; and iii) Riparian buffer strips as a multifunctional management tool in agricultural landscapes. The three workshops covered a wide variety of scientific presentations from example projects and areas around the member countries of the Action. In this paper we highlight a few case studies presented at the three workshops. The main outcome of the working group 4 discussions show that we have a lack of evidence on the outcome of implementing different mitigation measures at specific critical locations within the landscape when we are monitoring overall effects at the catchment scale. This is due to a limited knowledge of the importance of biogeochemical processes (denitrification, phosphorus sorption/desorption, nutrient reservoirs and storages), hydrological time delays (so called 'lag effects') and side-effects such as pollution swapping taking place between the field and catchment scale. Consequently, observed trends in nutrient concentrations may therefore not give the catchment manager the entire true signal when statistical trend tests are performed based on monitoring data from catchment outlets. It is a crucial part of cost-effectiveness evaluation that our catchment observations (particularly when part of national regulation/evaluation of costly RBMPs) are sufficiently robust to determine if change has or has not occurred, or evaluate signs of improvement, as a result of mitigation efforts. Robust monitoring includes the correct ecological indicators applied at correct spatial and temporal scales.

**Introduction**

As part of the EU Water Framework Directive (WFD), EU Member States shall collect and maintain information on the type and magnitude of significant anthropogenic pressures on water bodies, leading to ecological impacts. Among these anthropogenic pressures is the loss of nutrients from diffuse sources, since excess nutrient loadings into rivers, lakes, reservoirs and estuaries lead to eutrophication which, through algae growth, can severely impact freshwater and marine ecosystems (e.g. KRONVANG *et al.,* 2005a).

River Basin District Authorities (RBDA) in Europe have to conduct an analysis for each catchment, based on existing data such as land use, pollution sources and water monitoring data. Such an analysis is often performed in a stepwise manner following, for example, the DPSIR concept (Driving forces, Pressures, State, Impacts and Responses) (Fig. 1).

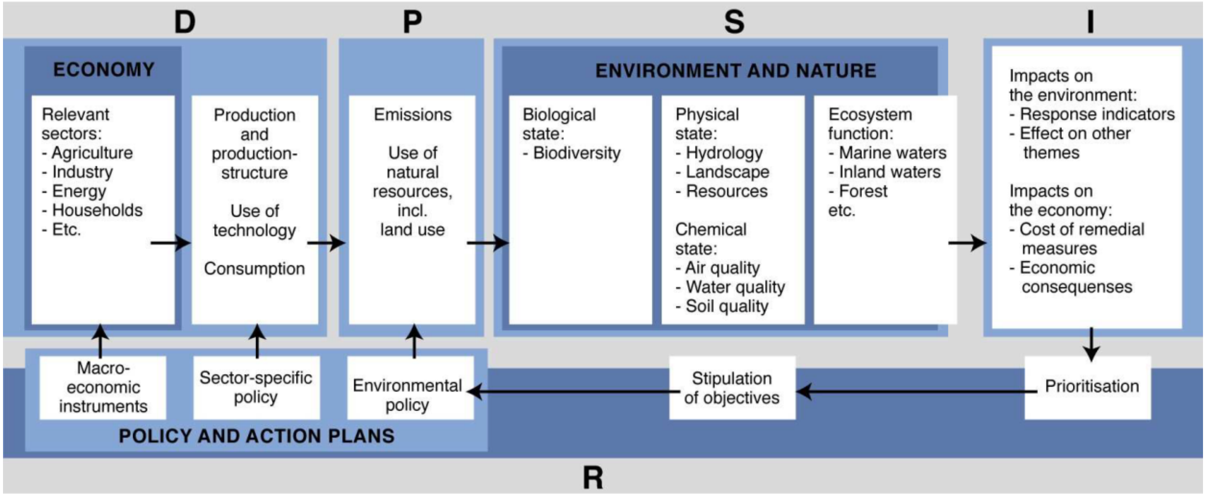


*Figure 1.* Diagram of the DPSIR concept.

In cases where the DPSIR analysis reveals that nutrient loss from diffuse sources causes water bodies to fall short of the WFD requirements for good ecological quality, RBDA and catchment managers have to make responses through the development of River Basin Management Plans (RBMP). Such RBMP's will have to incorporate different general and targeted mitigation measures that have to be adopted for reducing diffuse nutrient pollution in order to make each water body meet a good ecological quality. Since both nitrogen (N) and phosphorus (P) losses to surface waters and groundwater are in many cases largely driven by agriculture, there is an urgent need in Europe to learn about the relationships between various general and targeted mitigation measures hitherto adopted and their resultant effect at River Basin scale (e.g. KRONVANG *et al.,* 2005b). There is a great need for improving our common understanding of uncertainties related to implementation of different mitigation measures at River Basin scale in different regions of Europe, caused by different mechanisms such as inertia caused by time lags in responses (KRONVANG *et al*., 2009), nutrient retention (HEJZLAR *et al.,* 2009), climate change impacts (JEPPESEN *et al.,* 2011).

The River Basin District Authorities have to fulfil the requirements of monitoring surface water and groundwater under the WFD, by establishing a monitoring network and adopting sampling protocols designed to provide a coherent and comprehensive overview of the ecological status of water bodies in River Basins. The monitoring programmes implemented should, however, also be able to detect the benefits of mitigation measures implemented to combat diffuse nutrient losses in River Basins throughout Europe. Such monitoring requirements demand that harmonised sampling protocols are developed that can cope with the wide ranges in catchment geology and hydrology existing in European River Basins, in order to enable a precise and reliable documentation for the short term and longer-term effects of different mitigation measures implemented.

The aim of Working Group 4 (WG4) under the EU Cost 869 Action was to exploit the existing European-wide knowledge on combating nutrient pollution from diffuse sources in River Basins, gained through demonstration projects implementing general and targeted mitigation measures. The WG4 was tasked with contributing to the development, implementation, monitoring and assessment of River Basin Management Plans across Europe, by learning from the experience gathered in EU Directives (e.g. Nitrate Directive), national Action Plans, supra-national River Basin Action Plans (e.g. Conventional areas as the Baltic Sea Comission (HELCOM) and The North Sea Commission (OSPARCOM)) and regional river basin Action plans for combating nutrient pollution, conducted at different scales during the last 1-2 decades.

**Meetings held under WG4**

A first WG4 meeting was held on 18-22 May 2008 in Waidhofen, Austria. The topic of the meeting was: '*Evaluation of Projects in Example Areas across Europe'.* A total of 22 oral presentations on different projects and 6 posters were presented during the workshop. On Wednesday the 22nd of May 2008 a trend analysis session was held where participating countries discussed the outcome of analysing nutrient concentration data as a method for observing results of mitigation options implemented at e.g. catchment scale.

A second WG4 meeting was held on 24-26 June 2009 in Nottwil, Switzerland. The topic of the meeting was: '*Evaluation of projects in example areas: The Swiss Midland Lakes'.* A total of 17 oral presentations on different example projects and 13 posters were presented during the workshop.

A third WG4 meeting was held on 25-28 April 2010 in Ballater, Scotland. The topic of the meeting was: '*Riparian buffer strips as a multifunctional management tool in agricultural landscapes'.* A total of 24 oral presentations on buffer strip functioning and 8 posters were presented during the workshop.

**Results and discussions**

Lake Sempach – an example area for combating nutrient pollution

A special case of longer-term nutrient management was presented and discussed at the 2nd meeting in Switzerland. The lake area is 4.4 km$^2$, mean lake depth is 44 m and the catchment area is 61.9 km$^2$. Land use in the catchment is dominated by grassland (60%), followed by arable land (16%), forest (16%) and urban areas (8%). The livestock density in the catchment is 2.4 LU's per ha and a total of 13,500 inhabitants live in the catchment.

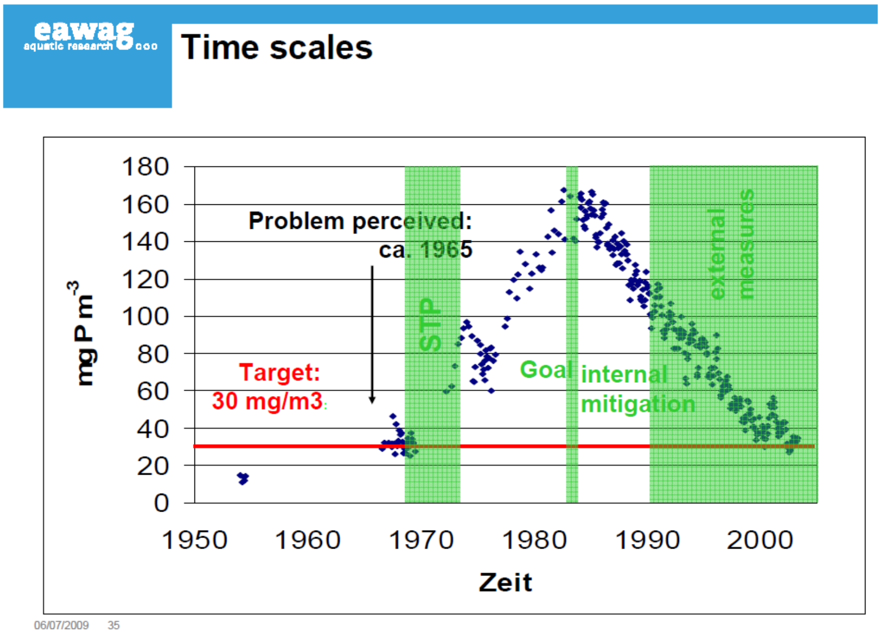


*Figure 2*. History of development in phosphorus concentration in the Lake Sempach, Switzerland and the different management strategies adopted.

Signs of eutrophication have been observed in the Lake Sempach since the 1960s (Fig. 2). Since that time, the implementation of mitigation measures tackling urban and agricultural sources of phosphorus input has been closely linked to research on the causes behind eutrophication and possibilities for remediation. Agricultural losses of P were considered of minor importance in the 1970s based on the available data. However, monitoring in the lake tributaries revealed strongly increased P-loads by the mid 1980s. This observation fostered investigations clarifying the role of different processes causing the high P export from the grassland dominated catchment.

The mitigation history of the catchments is:

- 1968 - 1976: Construction of Waste Water Treatment Plants
- 1983: Foundation of the cooperation of lake municipalities
- 1984: Massive fish kill, Start artificial oxygenation & water circulation
- 1986: P ban in detergents (entire Switzerland)
- 1988: Max. stock density at 3 LSU/ha
- 1993: Revision Swiss agricultural policy
- 1997: Oxygenation stopped
- 1999: Targeted project in agriculture (Art. 62a)

Today, the P concentrations have been reduced below a critical concentration of 0.03 mg/L. Understanding the rate of decrease in observed P concentrations can only be obtained by considering the interplay between lake-internal P cycling and reduced P input due to external mitigation measures.

The lessons learned from the lake Sempach study regarding controlling nutrient pollution are:

- Treatment of sewage from point sources is an important first step in nutrient management.
- Lake-internal effects on P-dynamics are important to understand (model or mass-balance).
- Any change in lake-internal processes may be critical as lake status may be misleading as measure of success in agriculture.
- Mitigation efforts in agriculture are a success story. Need to continue efforts in agriculture to prevent future losses.
- Time scales are important in P mitigation, in the example of lake Sempach it took more than 40 years to reach the P threshold set for the lake to obtain a good ecological condition.

Another outcome of the workshop on combating nutrient pollution from diffuse agricultural sources is the importance of working together with the stakeholders. Thus, it is crucial to involve all stakeholders from day 1 and through the whole process, and to conduct a holistic process where production, nature and environmental goals are settled at the same time. This is because such an approach creates cost-effective benefits (synergy).

<u>Trend analysis of stream water flow and nutrient data</u>

Water flow, nitrogen and phosphorus concentration data from nine countries: The Netherlands (1), Denmark (2), Sweden (1), Norway (2), Scotland (7), Czech Republic (1), Germany (3), Lithuania (3) and Austria (1). Time series from a total of 21 agricultural catchments were submitted to NERI, Denmark for statistical analysis of trend. The statistical trend analysis was undertaken using Kendall's seasonal trend test with correction for serial correlation. This test is robust non-parametric site-specific statistical tests for monotone trends and is described in Appendix 1.

The outcome of the trend analysis was delivered to the delegates from each country before the workshop. Each delegate presented the results at the trend workshop for their catchments and discussed why the datasets showed either upward or downward trends linking to information from the catchment on development in point sources or non-point sources and catchment management. The main outcome of the trend analysis on water flow (daily discharge), nitrogen concentrations and phosphorus concentrations is shown in Table 1. Most of the catchments experienced no trend in water flow (81% of the studied catchments). Most catchments experienced downward trends in nitrogen concentrations (total N: 46% and nitrate-N: 40%). Phosphorus concentrations showed both downward (total P: 25%, SRP: 13%) and upward trends (total P: 20%; SRP: 33%) (Table 1).

*Table 1*. Number of catchments experiencing a significant ($P<0.05$) trend or no trend.

| Trend | Water flow | Total N | Nitrate-N | Total P | SRP |
|---|---|---|---|---|---|
| Upward | 2 | 4 | 4 | 4 | 5 |
| Downward | 2 | 6 | 8 | 5 | 2 |
| No trend | 17 | 3 | 8 | 11 | 8 |

Important factors that might influence observations of trends can be outlined as:

- Measures in some cases implemented before start of monitoring.
- Changes in management practices (or external influences such as cereal commodity prices) counteract effects of implemented mitigation measures. Huge variability in seasonal and inter-annual hydro-meteorological conditions (climate change causing increased nutrient losses may partly override the effect of mitigation strategies).
- Time lags – travel time for nutrients from source to recipient?
- Influence of sinks – nutrient retention in groundwater and surface water bodies?
- Wrong trend test method?
- Not enough sampling conducted?

*Table 2.* Description of the Odderbæk and Lillebæk catchments in Denmark

| | Odderbæk | Lillebæk |
|---|---|---|
| Period tested | 1989-2006 | 1989-2006 |
| Catchment size | 1140 ha | 470 ha |
| Main soil type | coarse sandy soil (72%) | loamy sand soil (86%) |
| Deeper soils | mixture clay and sand – Groundwater dominated | clay and sand |
| Average annual precipitation | 872 mm | 831 mm |
| Agricultural land, dominated arable | 98% | 89% |
| | Land is partly tile drained | Intensively tile drained |
| Animal density | 1.36 Livestock units (2003) | 1.05 Livestock units (2003) |

An example of the outcome of trend analysis tests is given from the two Danish streams. The description of the two streams and catchments are shown in Table 2 and 3. The time series of nitrate-N concentrations analysed for trends are shown in Figure 3.

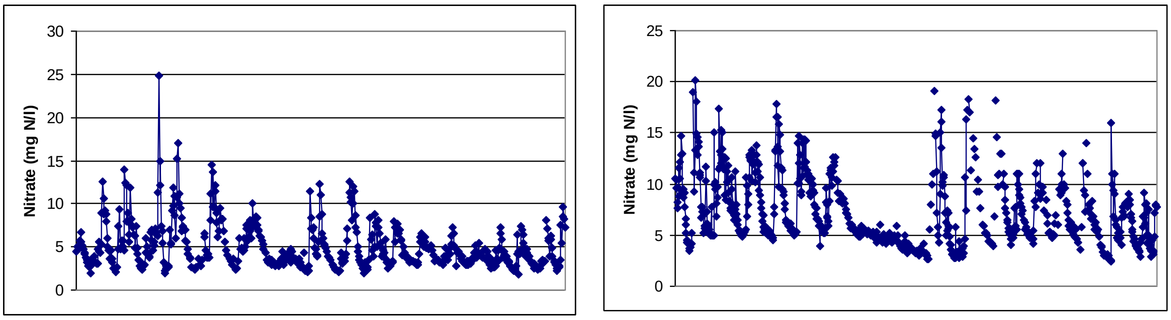


*Figure 3.* Observed concentrations of nitrate-N during the period 1989-2006 in the two streams Odderbæk and Lillebæk, Denmark.

The outcome of the trend analysis for the two Danish streams is shown in Table 4 and 5. Significant downward trends were detected for total N and nitrate-N in both streams as a result of the mitigation measures adopted in agriculture which reduced the N surplus in both catchments. Although the reduction in N surplus was greatest in the Odderbæk catchment the response as measured from the trend analysis was less pronounced as opposed to the results from the Lillebæk catchment. These findings may be a result of the contrasting hydrology in the two catchments being groundwater dominated in the Odderbæk and more linked to tile drain flows in the Lillebæk catchment. The difference can be seen to affect the nitrate-N concentration dynamics in the two streams being more fluctuation in the Lillebæk than in the Odderbæk (Fig. 3).

*Table 3.* Agricultural statistical information on Nitrogen cycling in the two catchments based on annual interviews with farmers.

| | Odderbæk | Lillebæk |
|---|---|---|
| | Kg N ha$^{-1}$ | |
| Average input of N in 1991 | 312 | 220 |
| Average harvested N in 1991 | 150 | 140 |
| Net input in 1991 (surplus) | 162 | 80 |
| Average input of N in 2003 | 230 | 169 |
| Average harvested N in 2003 | 140 | 100 |
| Net input in 2003 (surplus) | 90 | 69 |
| Reduction in surplus 1991-2003 | 72 | 11 |
| Reduction as percentage | 44 | 14 |

*Table 4*. Results of the trend analysis performed on time series from the Odderbæk stream.

| | Discharge | Total N | NO3 | Total P | SRP |
|---|---|---|---|---|---|
| Annual trend | 0.7 l/s ** | - 0.071 mg N/l *** | - 0.073 mg N/l *** | NST (+) | NST (+) |
| Whole trend 1989-2006 | 12.6 l/s (16%) | - 1.28 mg N/l (26%) | - 1.31 mg N/l (33%) | | |

NST: No Significant Trend. *: significant at 0.05; **: significant at 0.01; ***: significant at 0.001

*Table 5.* Results of the trend analysis performed on time series from the Lillebæk stream.

| | Discharge | Total N | NO3 | Total P | SRP |
|---|---|---|---|---|---|
| Annual | NST | - 0.181 mg N/l *** | - 0.166 mg N/l *** | -0.0026 mg P/l *** | - 0.0007 mg P/l * |
| 1989-2006 | | - 3.26 mg N/l (40%) | - 2.99 mg N/l (37%) | -0.047 mg P/l (22%) | -0.013 mg P/l (11%) |

NST: No Significant Trend. *: significant at 0.05; **: significant at 0.01; ***: significant at 0.001

Lag times and/or retention of nitrogen are thus playing a more important factor in Odderbæk than in Lillebæk catchment. No significant trends were found for P in Odderbæk, whereas a significant downward trend was detected in Lillebæk (Table 4 and 5). The reason for not finding any reductions in P concentrations in the Odderbæk is probably that a positive P-surplus still prevails in catchments having a high livestock density. The reason behind the decrease in Lillebæk can possibly be ascribed to improved treatment of sewage water at single houses and farms in the catchment more than any changes in diffuse P losses from fields. Trend analysis certainly has a place in evaluation of mitigation effectiveness. However, the issues highlighted above suggest that a more experimental approach, such as a paired control and intervention catchment approach would be more desirable and scientifically valid (as accounts for inter-annual hydrological change etc).

Buffer strips as mitigation option

It is envisaged that riparian buffer strips can have immediate and long-term benefits for diffuse pollution control, biodiversity and communities. However, the basis on which these buffers are designed and placed in landscapes requires guidance based on scientific understanding and we need to assess and maximise their effectiveness and lifespan in respect of these multiple linked benefits. The workshop held on buffer strips focused on bringing together scientists and catchment practitioners from across the EU to further our understanding of riparian buffer functioning in different situations so that practical guidance may be developed as to their use, design, management and limitations. A specific meeting assessing buffer strips is timely. Buffer strips appear in 70% of the RBMP across the EU and seem to be a readily adopted measure with perceived effectiveness. There are a number of upcoming pieces of EU legislation which will provide tighter restrictions on farming activities at watercourse margins. Research and shared understanding is necessary if the wide-scale uptake of buffer strips into riparian management is to be made most effective in terms of achieving multiple environmental functions (STUTTER *et al.,* 2012).

The workshop discussed several topics around buffer strips and a summary of the findings and recommendations for function, design and management are given in Table 6 and 7. Awareness raising through communication with the farming community is a big effort and needs to be done effectively. Often the people with whom it is most important to communicate (those with the poorest practices) will not negotiate. Wider groups of people need communication with (and between) to better promote general environmental awareness and specifically riparian management. These groups could be farmers, children in farming areas, farming cooperatives, up to supermarket food buyers. Good riparian management may be a means for a farmer to visibly improve an accessible area of the landscape and promote a better environmental image for farming. In turn, we (the science to policy community) need to be clear at communicating what we expect from buffers. It is unlikely that buffers will achieve common goals in all circumstances. We need a system for analysing problems in the local landscape and setting appropriate goals. Maybe buffer ‘strips’ need a new name as ‘strip’ expresses a conflict in land between farming and the environment. We should consider the use of the term ‘buffer zone’ instead.

*Table 6.* Function of buffer strips as discussed at group meetings at the workshop in Ballater, Scotland.

| Function | Issues |
|---|---|
| *Water quality* | Insufficient knowledge at present, site specific factors important, uncertainty in data and models needs to be communicated, pollutant swapping (e.g. GHGs), insufficient knowledge of N dynamics, problems in long-term nutrient storage (esp. P) leading to leaching, interactions with vegetation management not known, timing and nutrient form of leaching to watercourses important for eutrophication, need more studies especially at catchment scales |
| *Habitat improvement, biodiversity* | There is a conflict between nutrients and biodiversity services, perhaps two zone model buffer zone (closest to field) then eco-zone (adjacent to river), need for better modelling, maybe separate farming and areas of nature value to alleviate conflicts? |
| *Shading* | Plant tree species to encourage shading and leaf litter/woody biomass inputs, useful to combat temperature increase due to climate change |
| *Flow capture* | Useful reconnection of watercourses with their floodplains, fulfils multiple policy objectives, promotes seed dispersal of land plants, linked to wetlands and their potential as bioreactors, does sediment returned to the floodplain bring contamination issues (downstream of WWTP, urban areas etc)? |
| *Carbon sequestration* | Buffers have greater topsoil C contents, perhaps interactions with tree planting or leaching as DOC. Does soil C availability and inputs to watercourses promote nutrient processing (e.g. terrestrial or in-stream denitrification)? |
| *Biomass production* | May offset economics of land taken from farming, timber production or biofuel crops are examples. Could products be harvested without degradation of the buffer? |
| *Landscape Diversity* | Need to replace much of what has been lost through agricultural intensification over last 50 years, tools are required for better landscape planning, need to link riparian management with wider catchment management |
| *Cultural services* | Hunting species are important (fishing, game birds), public access and recreation, community education (e.g. school groups) |

*Table 7*. Design and Management options for buffer strips as discussed at group meetings during the workshop in Ballater, Scotland.

| *Design* | Links to function | What do we need to know? |
|---|---|---|
| Dimensions, width | Appropriate width depends on purpose, but is not fixed<br>Biodiversity benefits from widening buffer, DP mitigation may reach an optimal width.<br>There is a need to balance width and design to the specific landscape. | Do you have to buffer the buffer to achieve biodiversity or DP goals for narrower buffers?<br>What are the economics of different buffer widths?<br>Is a narrow, continuous buffer better than a wider, discontinuous buffer at hotspots? |
| Vegetation type | Trees provide leaf litter and shading functions, grasses provide water percolation and erosion trapping functions, shrubs may provide cover for birds.<br>There is a conflict between nutrient enrichment and the type of plants that will grow. | The optimal mixture of vegetation between grasses, trees, hedges etc.<br>Need appropriate set of indicators for 'valued' plant species. |
| Fencing | Necessary to exclude animals that cause bank erosion | Is temporary fencing better than fixed? |
| Soil amendments | Reactive amendments (e.g. Fe oxides) could be used to help retain phosphorus<br>Could organic matter amendments increase N transformations? | What is the buffer lifespan for P retention with/without amendments? |
| Topographic and flow structures | Retention ponds, or bunds, leaky barriers may be useful | What are the lifespans before needing emptying? |
| *Management* | | |
| Nutrient off-take | This can improve the lifespan of the buffer by removing nutrients | How should it be done to be most (cost) effective? |
| Grazing | Not in the first year of buffer life, beware of compaction and erosion by animals, requires careful management | |
| Sediment | Sediment could potentially have high yield of P so part of nutrient recycling<br>Sediment may be contaminated though | Need to establish the costeffectiveness of recovering the soil and nutrients and putting it back to the land |
| Vegetation removal | Removal of cuttings may stop P leaching | |
| Monitoring | There should be some form of observations to establish achievement of goals and hone management | Should we be checking soil properties, biological factors, or water quality? |

Perhaps there are not sufficient public funds to rely on incentives alone, so legislation and education have their places. However, farmers should be compensated for providing important ecosystem services that are costly, or disadvantage their businesses. We question whether these should be paid on the basis of implementation, or results, though the latter are more difficult to judge and administer a system around. Are current economic instruments appropriate to offset environmental measures such as buffer strips? Allowing activities that produce a profit from riparian areas (biofuel crops, timber, hunting revenue) may help close any gaps. A further aspect could be the improved status of the farmer in the community as a 'land steward' in addition to actual payments.

Three main issues of concern when dealing with buffer strips were raised during the discussions in WG4:

A) Legal issues

The legal framework of buffer strip implementation seems to be divided into compulsory and voluntary part. In many EU countries compulsory measures exist to protect open water. However, the extent of the compulsory part of buffer strip implementation varies between countries and is in a range of between 0.5 m (the Netherlands) and 20 m (Austria). In addition, buffer widths differ between types of waters (lakes or streams) and sizes of water course. In some countries, the extent of mandatory buffer strips is also based on additional constraining parameters such as slopes or adjacent field management. As an example: in Austria minimum buffer strip requirements for lakes are either 20 m when lakes are bigger than 1 ha or 10 m in case lakes are smaller than 1 ha. Along flowing waters the minimum size of buffer strips is 10 m in case of a mean slope of bordering fields of larger than 10 % and 5 m when mean slope is less than 10 %. Additional regulations apply in case of growing plants with late development such as maize or sugar beets (Aktionsprogramm 2008). In addition to the mandatory regulations many countries offer extended buffer strip options on a voluntary basis. Some countries do not consider buffer strip implementation at all. An overview about regulations in some of the European countries is given in DWORAK *et al*. (2009).

B) Problems of implementation

A major problem of implementation seems to be the conflicts of interest in designing buffer strips. A major conflict which arises is the different views between environmental protection issues and agricultural production issues. This is quite obvious and well known. However, in addition different views of how an optimum buffer strip should be designed and managed exist between the different environmental lobbies. The examples discussed were the implementation of the energy crop Miscanthus on buffer strips to generate additional income for landowners and increase implementation. This is seen as critical by river ecologists as the roots of Miscanthus may be transported easily by the flowing water and in such a way invade other parts of the landscape. Buffer strips are also favoured by environmentalists to increase biodiversity. Optimising buffers for biodiversity may entail a very different style and extent of management than for diffuse pollution purposes. Hydrologically seen, buffer strips are the final means of 'filtration' for surface runoff as it enters the stream. Therefore, the sediment load of the surface runoff would already be high. Consequently, retention of any dissolved or particulate matter eventually will lead to nutrient accumulation within the buffer strip, which finally may act as a source instead of acting as a sink. Removal of plants might therefore help to maintain effectiveness of buffer strips in the long term (BEDARD-HAUGHN, *et al.,* 2004; OSBORNE and KOVACIC, 1993). An additional consideration also in this context is pollution swapping (STEVENS and QUINTON, 2009) which could lead to additional undesired effects.

More practical problems discussed were that management of buffer strips needs careful consideration. Many times farmers use grassed buffer strips to turn their machinery. Because the soil of buffer strips does not get any actual soil management, this may easily lead to permanent compaction which in turn will reduce the buffer strip effectiveness. In addition linear flow paths (cow tracks, machinery track lines) may lead to a concentration of flowing surface water, thereby reducing its effectiveness drastically. In addition natural depressions may lead to concentration of superficially running water. Therefore for any granting scheme flexibility to allow for local adaption would be highly desirable. To add to this, there might also be measures in support of buffer strips which are not proposed in any scheme at present, such as use of straw balls to retain surface water or increasing soil height at specific sites to create small retention structures.

Is it necessary to fence buffer strips? This was another topic discussed. In case of grazing animals alongside of waterways this would be desirable, however, the costs of fencing seem to constrain an effective implementation of fences.

C) Effectiveness of buffer strips including cost effectiveness

Since long term discussions about effectiveness of buffer strips are continuing. A variety of diverging results on the topic exist. Table 8 lists a few studies to give an overview about measured effectiveness for phosphorus/sediment and additional factors that may influence the outcome of implementing buffer strips. It is obvious, that nothing such as a mean effectiveness exists and depending on local conditions very different results may be obtained from implementing buffer strips.

Therefore the question arose as to the evaluation of buffer strips. It was agreed that a true evaluation for areas larger than the plot scale (=watershed scale) would be very difficult to carry out. This is because of two main reasons, a) the difficulty to exactly pin changes at particular fields/sites in a watershed to changes observed at an outlet station and b) the temporal delays between implementation of a field/site specific measure and the measurable response at the catchment outlet. One possibility to overcome large scale evaluation problems is upscaling of plot results and/or modelling. As an example, the reduction of phosphorus input into Danish surface waters from establishing 10 m wide buffer strips was calculated at national scale to be 5-7% based on experimental data on the P reduction for pathways such as surface runoff, leaching and bank erosion (KRONVANG *et al.,* 2005b).

*Table 8:* Overview on effectiveness of vegetated buffer strips in small scale studies

| Reference | length (m) | slope (%) | Soil type | Retention (%) |
|---|---|---|---|---|
| DILLAHA et. al. (1989) | 4.6 – 9.1 | 11 - 16 | silty loam | 53 - 98 |
| MAGETTE et. al. (1989) | 4.6 – 9.2 | 2 - 4 | sandy loam | 66 - 82 |
| MUÑOZ-CARPENA et. al. (1993) | 4.3 – 8.5 | 5 - 20 | silty Loam, clay loam | 80 - 95 |
| DANIELS u. GILLIAM (1996) | 48 - 86 | 2.1 - 10 | sandy loam, clay loam | 55 - 82 |
| GHARABAGHI *et al*. (2000) | 2.4 - 19.5 | 5.1-7.2 | clay loam | 50 - 98 |
| SYVERSEN (2001) | 5 | 14 | silty loam | 55 - 80 |
| DOSSKEY et. al. (2002) | 9 - 35 | 2 – 3.8 | loam | 15 - 43 |
| ABU-ZREIG *et al*. (2003) | 2 - 15 | 2.3 - 5 | sandy loam | 65 - 91 |
| FOX *et al*. (2005) | 15 | 6.5 | sandy loam, silty loam | 73 - 99 |
| HELMERS *et al*. (2005) | 13 | 1 | silty loam | 61 - 87 |
| KUO (2007) | 4.1 – 13.4 | 2 – 4.3 | sand | > 96 |
| DOSSKEY *et al*. (2008) | 200 - 400 | 2. 10 | silty loam | 30 - 99 |

The outcome for reductions in N leaching was estimated to amount to 2500 tons N at the national scale in the same investigation. Attached to such kind of evaluation is the question of cost effectiveness of buffer strip implementation. To improve results of large scale modelling local inspections together with farmers could be undertaken.

**Conclusions and perspectives**

The objectives of WG4 were reached by hosting three scientific workshops in 2008, 2009 and 2010. During these workshops we established an overview of different general and targeted mitigation measures already implemented in different member countries. We found that there is a gap in our knowledge regarding the effects of implementing mitigation measures at scales larger than plot and field. Well documented demonstration studies such as Lake Sempach are very seldom found in the international literature. The outcome of implementing different mitigation measures at the catchment and river basin scale is therefore only known from upscaling and modelling which introduces a high uncertainty in the final estimates. Even worse is the lack of knowledge on processes linked to inertia of nutrients in catchments (removal processes, storage processes, time delays) that may introduce false evidences from monitoring of the effects at catchment outlets. A situation may often arise, when using a typical 'regulatory monitoring approach' that scientists and catchment managers are left unable to discern between three possible scenarios, namely that:

- monitoring was of insufficient resolution or design to detect change (ie. improvements may, or may not have occurred but couldn't be observed with any statistical significance or power)
- Catchment restoration actions were insufficient to make any improvements? (ie. monitoring correctly observed no improvements)

Inherent lags in the system meant that any beneficial changes were yet to manifest themselves? *(ie. actions were sufficient to begin to cause change but this was slow to manifest itself in observable improvements)* Demonstration studies at the catchment scale is very costly and we therefore urge countries wi*thin different European regions to establish research groups that work together for documentation of the effects of mi*tigation measures adopted at different scales. The River Basin Management Plans now established in most EU countries as part of the Water Framework Directive are an obvious working platform for the future capture of evidence at larger scales of mitigation measures implemented during the first plan period (2009-2015).

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Appendix 1

**Method for trend analysis**

Trend analysis of time series of nutrient concentrations and runoff at river stations in catchments was undertaken using Kendall's seasonal trend test with correction for serial correlation. This test is robust non-parametric site-specific statistical tests for monotone trends. It is robust towards missing values, values reported as "< detection limit", seasonal effects, autocorrelated measurements and non-normality (i.e. non-Gaussian data). The test was introduced in the papers HIRSCH *et al*. (1982) and HIRSCH and SLACK (1984) and has become a very popular and effective method for trend analysis of water quality data. The statistical trend method can analyse both seasonal and annual data and provide a trend statistic, *P*-value and an estimate of the annual increase or decrease in nutrient concentrations.

A trend analysis starts out with a time series plot (a graph showing observed concentrations versus time of observation) and a Box-Whisker plot (a graph showing the distribution of data for each calendar month). Such plots can give hints on possible trends, seasonality and extreme values.

Both total nitrogen and total phosphorus concentrations are highly depending on runoff. This substance-specific relationship can be modelled by the non-parametric and robust curve fitting method LOWESS (Locally Weighted Scatterplot Smoothing, Cleveland, 1979). The nutrient concentrations must be adjusted for runoff in order to minimise the impact from climate and to prevent a deterioration of the trend detection thereby increasing the power of the test. To remove the effects of runoff calculate residuals, i.e.

$$r = x - \hat{x}_{(LOWESS)},$$

where $\hat{x}_{(LOWESS)}$ is the estimated concentration from LOWESS and $x$ is the observed concentration. A time series plot of the residuals will reveal if the trend is still present in the adjusted values (residuals).

The trend method only operates with one value for each combination of season and year. Therefore an average value for the seasons with more than one observation is used. Let $r_{ij}$ denote the average value of all adjusted measurements in year *i* and season *j*. It is assumed that there have been measurement in *n* years and *p* seasons, i.e. $i = 1,2\ldots,n$ and $j = 1,2,\ldots,p$. In normal applications the number of seasons *p* per year was set to 12 one for each month of the year. Some of the $r_{ij}$'s can be missing if no measurement have been done in the relevant month and year.

The null hypothesis of the trend analysis is: for each of the *p* seasons the *n* data values are randomly ordered. The null hypothesis is tested against the alternative hypothesis: one or more of the seasons have a monotone trend. The trend test is done by calculating

$$S_g = \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} \operatorname{sgn}\left(r_{jg} - r_{ig}\right),$$

for $g = 1,2\ldots, p$, and where

$$\operatorname{sgn}(x) = \begin{cases} 1, & x > 0 \\ 0, & x = 0 \\ -1, & x < 0 \end{cases}.$$

If $r_{jg}$ and/or $r_{ig}$ is a missing value, then $\operatorname{sgn}\left(r_{jg} - r_{ig}\right) = 0$ per definition.

A combined test for all seasons (months) is done by first calculating

$$S = \sum_{g=1}^{p} S_g \,,$$

and

$$\operatorname{var}(S) = \sum_{g=1}^{p} \operatorname{var}\left(S_g\right) + \sum_{g,h:g\neq h} \operatorname{cov}\left(S_g, S_h\right).$$

The variance for $S_g$ under the null hypothesis can be calculated exactly by

$$\operatorname{var}\left(S_g\right) = \frac{n_g\left(n_g - 1\right)\left(2n_g + 5\right) - \sum_{j=1}^{m} t_j\left(t_j - 1\right)\left(2t_j + 5\right)}{18},$$

where $n_g$ is the number of non-missing observations in season $g$. In the formula for the variance of $S_g$ it is assumed that there are groups of observations with completely equal values, $m$ groups in total and in the $j$'th group there is $t_j$ equal values.

It is not possible under the null hypothesis to calculate the covariance between $S_g$ and $S_h$ exactly, but it can be estimated by (HIRSCH and SLACK, 1984)

$$\operatorname{cov}(S_g, S_h) = \frac{K_{gh} + 4\sum_{i=1}^{n} R_{ig} R_{ih} - n(n_g + 1)(n_h + 1)}{3},$$

where

$$K_{gh} = \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} \operatorname{sgn}\left[(r_{jg} - r_{ig})(r_{jh} - r_{ih})\right],$$

and

$$R_{ig} = \frac{n_g + 1 + \sum_{j=1}^{n} \operatorname{sgn}(r_{ig} - r_{jg})}{2}.$$

The term $R_{ig}$ is the ranking of $x_{ig}$ amongst all observations in season $g$, and all the missing values get the value $(n_g + 1)/2$ as ranking.

The test statistic for the aggregate test is

$$Z = \begin{cases} \dfrac{S-1}{(\operatorname{var}(S))^{\frac{1}{2}}}, & S > 0 \\ 0, & S = 0 \\ \dfrac{S+1}{(\operatorname{var}(S))^{\frac{1}{2}}}, & S < 0 \end{cases}.$$

The sign of *Z* indicates an increasing (+) or decreasing (-) trend. Both increasing and decreasing trends are interesting. The null hypothesis must be rejected if the numerical value of $Z$ is greater than the $(\alpha/2)$-percentile in the Gaussian distribution with mean 0 and variance 1. Here $\alpha$ stands for the significance level, which typically is 5%. At the 5%-level all *Z*-values numerically greater than 1.96 are significant. The reason for evaluating $Z$ in a Gaussian distribution is that under the null hypothesis, $S$ has a Gaussian distribution with mean 0 and variance

$\mathrm{var}(S)$ for $n \to \infty$. The Gaussian approximation is good if $n \geq 10$ (HIRSCH and SLACK, 1984). This means 10 years of data with one concentration measurement for each month.

The trend in each season can be tested by calculating

$$Z_g = \begin{cases} \dfrac{S_g - 1}{\left(\mathrm{var}\left(S_g\right)\right)^{\frac{1}{2}}}, & S_g > 0 \\ 0, \quad S_g = 0 \\ \dfrac{S_g + 1}{\left(\mathrm{var}\left(S_g\right)\right)^{\frac{1}{2}}}, & S_g < 0 \end{cases}.$$

The null hypothesis of no trend is rejected if the numerical value of $Z_g$ is greater than the $\left(\alpha/2\right)$-percentile in the Gaussian distribution with mean 0 and variance 1.

It is possible to calculate an estimate for the trend (a slope estimate) if one assumes that the trend is constant (linear) during the period and the estimate is given as change per unit time (year). HIRSCH et al. (1982) introduced Kendall's seasonal slope estimator, which can be computed in the following way. For all pair of residuals $\left(r_{ij}, r_{kj}\right)$ with $j = 1,2\ldots, p$ and $1 \leq k < i \leq n$ calculate

$$d_{ijk} = \frac{r_{ij} - r_{kj}}{i - k}.$$

The slope estimator is then the median of all $d_{ijk}$-values and is robust, if the time series has serial correlation, seasonality and non-Gaussian data (HIRSCH *et al.,* 1982). A slope estimate for each season can be calculated in the same way.

A $100(1-\alpha)$ % confidence interval for the slope can be obtained by the following calculations

- Choose the wanted confidence level $\alpha$ (1, 5 or 10%) and use

$$Z_{1-\alpha/2} = \begin{cases} 2{,}576, & \alpha = 0{,}01 \\ 1{,}960, & \alpha = 0{,}05 \\ 1{,}645, & \alpha = 0{,}10 \end{cases}$$

in the following calculations. For a normal application we use a confidence level of 5%.

- Calculate

$$C_\alpha = Z_{1-\alpha/2} \cdot \left(\mathrm{var}(S)\right)^{\frac{1}{2}}.$$

- Calculate

$$M_1 = \frac{N - C_\alpha}{2},$$
$$M_2 = \frac{N + C_\alpha}{2},$$

where

$$N = \frac{1}{2}\sum_{g=1}^{p} n_g\left(n_g - 1\right).$$

- Lower and upper confidence limits are the $M_1$th largest and $\left(M_2 + 1\right)$'th largest value of the $N$ ranked slope estimates $d_{ijk}$ .

Using the modified Van Belle and Hughes test for homogeneity (1984) one can test the homogeneity of the separate season trend test. This homogeneity test must be non-significant in order to use the combined trend test.

Time series of daily runoff values also has to be tested for trends. The same trend test as described above can be used on the measured runoff values. Slope estimates and confidence intervals are computed following the methods described above. If no significant trends are detected in the runoff time series, any significant trend in the concentration time series is said to be anthropogenic in origin.

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# EUROPEAN UNION'S STRATEGY FOR DANUBE REGION AND THE IMPLEMENTATION OF THE WATER FRAMEWORK DIRECTIVE

*István IJJAS*

Department of Hydraulic and Water Resources Engineering, Faculty of Civil Engineering, Budapest University of Technology and Economics, Hungary, 1111 Budapest, Műegyetem rkp.3. ijjas@vit.bme.hu

**Abstract**

The first part of the paper presents the background information on the Danube Basin and Region, the EU Strategy for Danube Region, the Danube River Basin Management Plan, the Water Framework Directive, and the significant Danube basin-wide transboundary water management issues.

The objectives, the scope and the structure of the Danube Strategy, the priority areas, the actions and projects are discussed in the second part of the paper with specific regard to nutrient control.

**Keywords:** river basin management plan, macro-regional strategy, nutrient control

## Introduction

The Danube River Basin (DRB) is the "most international" river basin in the world, covering the territories of 19 countries. Fourteen countries with territories greater than 2,000 km² in the DRB cooperate in the framework of the International Commission for the Protection of the Danube River (ICPDR).

The DRB is not only characterized by its size (over 100 million people, and fifth of the EU surface) and large number of countries, but also by its diverse landscapes and the major socio-economic differences that exist between the upstream and downstream countries.

The DRB has been the subject of many environmental investigations and studies funded by the involved countries and a wide range of organizations. This paper has been drafted using materials developed in the previous programs including:

- The Danube... for whom and for what, Equipe Cousteau (1992)
- Danube Integrated Environmental Study, Environmental Programme for the Danube River Basin, EC PHARE-Programme (1993-1994)
- Strategic Action Plan (SAP), Environmental Program for the Danube River Basin (1994)
- SAP Implementation Plan (1995)
- Danube Nutrient Reduction Program (1997 –1999)
- ICPDR Joint Action Plan 2000- 2005
- EC supported DABLAS program (specific reports in 2002 and 2004)
- EU WFD Danube River Basin Analysis (2005)

- UNDP/GEF Danube Regional Project (2001 – 2006)
- UNDP-GEF-ICPDR Danube River Basin Updated Transboundary Diagnostic Analysis Based on EU Water Framework Directive - Analysis Report (2006)
- Danube River Basin Management Plan (2009)

The key conclusions of this paper address the aspects of water management at the basin-wide scale with specific regard to nutrient control issues. The key source for this paper is the Danube River Basin Management Plan abbreviated as DRBMP (ICPDR 2009). Complementary information can be obtained from the national river basin management plans of the DRB countries.

The Ministerial Meeting of the DRB countries on February 16, 2010 adopted the Danube River Basin Management Plan (ICPDR, 2009), which outlines concrete measures to be implemented by the year 2015 to improve the environmental conditions of the Danube and its tributaries. The measures include the reduction of organic and nutrient pollution, offsetting negative environmental effects of man-made structural changes to the river, improvements to urban wastewater systems, introduction of phosphate-free detergents in all markets and effective risk management of accidental pollution. Further measures to restore river continuity for fish migration as well as the reconnection of wetlands will be also implemented. The plan addresses key requirements of the European Union Water Framework Directive (WFD).

The Danube countries have developed the DRBM Plan entailing measures of basin-wide importance as well as setting the framework for more detailed plans at the sub-basin and/or national level. Not all countries of the Danube Basin are EU Member States, but all the countries have agreed to adopt and implement the WFD.

**The main basin-wide environmental problems in the Danube River Basin**

The DRBM Plan identified four significant transboundary issues that are a priority for the Danube Basin and the impact of the Danube River on the Black Sea (ICPDR, 2009):

1) Nutrient Pollution – potentially leading to over enrichment by nutrients and eutrophic conditions. The main sources were identified as point source emissions (municipal wastewater and industrial discharges) and diffuse sources from agriculture.
2) Organic Pollution – potentially leading to low dissolved oxygen levels in the receiving water. The main sources were identified as inadequate wastewater treatment from municipalities and diffuse sources.
3) Hazardous substances – potentially leading to environmentally toxic conditions.
4) Hydro-morphological alterations – that have led to a loss of wetlands, negative impacts on natural aquatic conditions and present migration barriers for fish.

In the RBM plan of the Danube, for 2015 (and for the period lasting until 2027), the future visions of river basin level, and the chances of the implementation thereof, were laid down.

The major Danube basin-wide visions for surface waters are as follows (ICPDR, 2009):

1) Organic pollution: zero emission of untreated wastewaters into the waters of the DRB.
2) Nutrient pollution: balanced management of nutrient emissions via point and diffuse sources in the entire DRB that neither the waters of the Danube nor the Black Sea are threatened or impacted by eutrophication.

3) Hazardous substances pollution: no risk or threat to human health and the aquatic ecosystem of the waters in the DRB and Black Sea waters impacted by the Danube River discharge.

4) Hydro-morphological alterations:

    - Interruption of river and habitat continuity: they will be managed in such way that the past, current and future hydro-morphological changes of the river shall not hinder the migration and spawning of fish anywhere in the whole area of the DRB.
    - Disconnection of adjacent floodplains and wetlands: the floodplains and wetlands in the entire DRB will be re-connected and restored. The integrated function of these riverine systems ensure the development of self-sustaining aquatic populations, flood protection and reduction of pollution in the DRB
    - Hydrological alterations: they are managed in such a way, that the aquatic ecosystem is not influenced in its natural development and distribution
    - Future infrastructure projects: they are conducted in a transparent way using best environmental practices and best available techniques in the entire DRB – impacts on or deterioration of the good status and negative transboundary effects are fully prevented, mitigated or compensated.

The major Danube basin-wide visions for groundwaters are as follows (ICPDR, 2009):

1) Groundwater quality: the emissions of polluting substances do not cause any deterioration of groundwater quality in the DRB. Where groundwater is already polluted, restoration to good quality will be the aim.
2) Groundwater quantity: the water use is appropriately balanced and does not exceed the available groundwater resource in the DRB, considering future impacts of climate change.

During the 1970s and 1980s, the trophic status of the Black Sea, and particularly the Northwest Shelf increased dramatically. The ICPDR and the UNDP/GEF program have been agreed upon the following short- and long-term targets for the recovery of the Sea (ICPDR, 2009):

- Short-term: to avoid exceeding loads of nutrients discharged into the Sea beyond those that existed in 1997.
- Long-term: to reduce the loads of nutrients discharged to levels allowing Black Sea ecosystems to recover to conditions similar to those of the 1960s.

The key conclusions of the DRBM Plan on the implementation of basin-wide environmental objectives and visions are as follows (ICPDR, 2009):

- The assessment of the ecological status will be further developed.
- The implementation of the environmental requirements and an equal level of measures in Non EU MS would be sufficient to solve the basin-wide environmental problems.
- Measures identified regarding organic pollution will not ensure the achievement of the environmental objectives on the basin-wide scale by 2015. Significant further efforts will still be necessary.

- The situation in the DRB and the Black Sea regarding N and P will be improved but not ensure the achievement of the environmental objectives. The reduction potential for the agricultural sector is difficult to quantify due to uncertainties in the future economic development of this sector, mainly in the middle and lower DRB. More stringent urban waste water treatment obligations with N and P removal, limitations on P in detergents (P ban in laundry detergents and dishwasher detergents), coordinated measures on a wider scale to reduce the atmospheric deposition of N, the knowledge and understanding of the connection between Danube loads and the ecological response of the Black Sea are all needed to achieve the environmental objectives
- The environmental objectives will not be achieved in 2015 regarding hazardous substances. Further studies, more monitoring data and further measures will still be necessary.
- Significant further efforts for the next RBM cycles will be necessary to address the pressures from all hydro-morphological components. It is expected that 108 barriers will be made passable for fish, whereas 824 river and habitat continuity interruptions will remain by 2015. Remaining continuity interruptions will be addressed by 2021 and 2027. It is recommended that initial measures focus on defined ecological priority river stretches.
- Additional restoration measures will be taken beyond 2015 to reconnect the adjacent floodplains and wetlands.
- A number of future infrastructure projects are being planned in the Danube Basin and these projects, if completed without attention to river alterations, could cause new pressures or make existing pressures on water status worse. Measures will be taken to address future infrastructure projects to reduce or prevent their potential impacts on good ecological status or good ecological potential.
- More investigations are needed on the significance of other relevant issues as the quality and quantity of sediments, invasive species, water quantity issues and climate change.
- Future infrastructure projects need to be 'climate proof' or coherent in their approach and provide flexible management tools.
- A sediment balance for the Danube Basin has to be developed, including identification of possible consequences due to climate change. Downstream of dams, the loss of sediments and the deepening of the riverbed require an artificial supply of material or other engineering measures to stabilize the riverbed.

**The EU Strategy for the Danube Region (EUSDR)**

The EU Strategy for the Danube Region (EUSDR) is a macro-regional strategy adopted by the European Commission in December 2010 and endorsed by the European Council in 2011. The Strategy was jointly developed by the European Commission, together with the Danube Region countries and stakeholders, in order to address common challenges together.

On the website of the strategy (www.danube-region.eu) you can find out about the latest developments of the strategy, its 11 priority areas and main actions and projects, as well as information about existing funding opportunities.

The Strategy is defined in a Communication (COM(2010)715, 2010), accompanied by a detailed Action Plan (European Commission, 2010), which presents the operational objectives and actions and demonstration projects of the EUSDR.

The Strategy is not about funding, it is about closer cooperation. There are no new EU funds, no new EU legislation, and no new EU structures, but a focus on closer synergies between authorities at all levels to maximize the impact of actions and funding.

The strategy addresses four main objectives, or “Pillars”:

1) Connecting the Region
2) Protecting the environment
3) Building prosperity in the Danube Region
4) Strengthening the Danube Region

These four pillars are translated into 11 priority areas (P1 – P11) coordinated by the participating countries, representing the main areas where the macro-regional strategy can contribute to improvements (Table 1).

*Table 1.* Priority areas of EU Strategy for the Danube Region

| Priority Area | Countries in charge of coordination |
|---|---|
| P1 Mobility and intermodality | Inland waterways: Austria, Romania<br>Rail, road and air: Slovenia, Serbia |
| P2 More sustainable energy | Hungary, Czech Republic |
| P3 Culture and tourism, people to people | Bulgaria, Romania |
| P4 Water Quality | Hungary, Slovakia |
| P5 Environmental risks | Hungary, Romania |
| P6 Biodiversity, landscapes, quality of air and soils | Germany (Bavaria), Croatia |
| P7 Knowledge society (research, education and ICT) | Slovakia, Serbia |
| P8 Competitiveness of enterprises | Germany (Baden-Württemberg), Croatia |
| P9 People and skills | Austria, Moldova |
| P10 Institutional capacity and cooperation | Austria (Vienna), Slovenia |
| P11 Security and organized crime | Germany, Bulgaria |

The second pillar has 3 priority areas:

- P4 Water Quality
- P5 Environmental risks
- P6 Biodiversity, landscapes, quality of air and soils

The objective of the P4 priority area is „to restore and maintain the quality of waters” (implementation of WFD and the measures of river basin management plans). The major objectives of a few actions and projects of this priority area is the control of nutrient loads.

The key actions of P4 are as follows:

1) Implementation of the Danube River Basin Management Plan

2) Cooperation at sub-basin level

3) Development of information systems

4) Development of wastewater treatment facilities

5) Establishment of buffer strips along the rivers to retain nutrients

6) Cooperation between authorities responsible for agriculture and environment

7) Limitations of phosphates in detergents

8) Treatment of hazardous substances and contaminated sludge

9) Control of substances that are considered problematic

10) Reducing water continuity interruptions

11) To limit water abstraction

12) Exchange of good practice in integrated water management

13) Safeguarding of drinking water supply

14) Integrated Coastal Zone Management

The major criteria for the selection of the actions and example projects should focus on the

- issues of basin-wide importance,
- issues based on the Danube River Basin Management Plan adopted by the Ministerial Meeting of the Danube River Basin countries,
- issues that require a basin-wide perspective and cooperation between each or most of the countries sharing the whole basin,
- issues that require inter-ministry or inter-sector coordinating mechanisms and integration of different policies,
- issues that demonstrate immediate and visible benefits for the people of the Region, that have an impact on the macro-region (or a significant part of it),
- projects which promote sustainable development and cover several regions and countries;
- projects which are coherent and mutually supportive, creating win-win solutions and
- projects which are realistic (technically feasible and with credible funding).

The EU Strategy for the Danube Region (EUSDR) "follows the footsteps of the EU Strategy for the Baltic Sea Region and builds on its good practices". The parallels of the water resources management aspects of the EU Strategies for the Baltic Sea Region and Danube Region (Table 2), and what to learn from the Baltic Sea Strategy should be considered.

*Table 2.* Some of the parallels of the Baltic Sea Region and the Danube Region Strategy

| Aspects | Baltic Sea Region | Danube Region |
|---|---|---|
| Number of participating countries | 11 | 14 |
| Number of EU Member States | 8 | 8 |
| Catchment area, km2 | 1 739 000 | 830 000 |
| Number of international river basin districts | 14 | 1 |
| Major legal tools | EU Water Framework Directive<br>EU Marine Strategy Directive | EU Water Framework Directive |
| Type of water management | Integrated Marine Basin Management and Integrated River Basin Management | Integrated River Basin Management |
| Number of Priority Areas | 15 | 11 |

**Conclusions**

The water-actions of EUSDR, the coordinated improvement of the status of waters will

- conserve the water resources for the future water uses,
- contribute to sustainable regional development,
- ensure the preservation and the restoration of the ecosystems,
- reduce the negative impacts on the Black Sea.

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# NP TURNOVER STUDIES ON EUROPEAN AND ON DANUBE BASIN LEVELS

*CSATHÓ, P.*[1] *– RADIMSZKY, L.*[2] *– NÉMETH, T.*[3]

[1] Research Institute for Soil Science and Agricultural Chemistry of the Hungarian Academy of Sciences, Budapest, Hungary; e-mail: csatho@rissac.hu

[2] Research Institute for Soil Science and Agricultural Chemistry of the Hungarian Academy of Sciences, Budapest, Hungary

[3] Hungarian Academy of Sciences, Budapest, Hungary

**Abstract**

According to Webster's Dictionary (1961), the description of "Union" is: "A uniting into a coherent and harmonious whole". This paper deals with the question whether the plant nutrition practice within the 27 EU countries fits this description of "Union", or not. The answer to this question is distressing. Instead of a trend towards equalization in surface NP balances and soil NP status in the 27 EU countries, a further and accelerated polarization has happened in the last 15 to 20 years resulting in severe environmental threats in some of the former 15 EU countries, especially in Belgium and the Netherlands, and causing severe agronomic, social, and rural development problems in most of the new 12 EU countries. The Nitrates Directive seems to be ineffective in stopping the disadvantageous trends and moving them into the right direction.

In the opinion of the authors, there is a need for a paradigm shift in the EU agro-environmental protection legislation. As a summary, according to the authors' opinion, in order for the aims of the various EU directives, strategies, policies and the new SPS system to be fulfilled, there must be a complete restructuring of the EU's livestock distribution, its import-export and price policy of agricultural goods.

Another study was conducted on the Danube River Basin level to estimate the contribution of the various factors (Population, Industry, Agriculture, Background) to surface water NP loads. This research was done under the framework of the Integrated Danube Research Program, funded by PHARE. Due to the introduction of untreated sewage directly into surface waters, the NP load contributed by population waste was especially high in Central Europe in the early 90s. The steps taken by the EU to protect surface waters have thus led to a dramatic reduction in point-source pollution caused by the NP contained in sewage. The same strict regulations should be inaugurated in the Western European countries and NUTS-2 regions with the highest livestock densities in order to diminish excessive diffuse NP loads into surface and subsurface waters effectively.

**Key words:** Agricultural NP loads, NP balance, groundwater nitrate contamination, soil P supply, polarization within EU, optimal livestock density

## Introduction

Nutrient balance, especially those of nitrogen and phosphorus, are important environmental indicators. The magnitude of mineral and organic NP use has a major effect on NP balance. Therefore, it is essential to investigate the major factors affecting them, such as per capita GDP and population density.

The aim of this study is to evaluate factors affecting the intensity of organic and mineral NP application per agricultural land unit, and to compare environmental NP balance –using the OECD methodology – and soil P supplies of the Western European former EU15 countries and new EU12 member countries in Central and Eastern Europe (OECD, 1997).

As a synthesis of a literature survey, Johnston and Steén compared the soil P supply data of Western European countries, most of them long-term EU members (STEÉN, 1997). CSATHÓ *et al*. (2007) have done the same for Central and Eastern European countries, allowing comparisons to be made between N balances and groundwater nitrate contamination, as well as P balances and soil P supplies within and between these two main European regions, i.e. the former EU15 and the new EU12 countries. Also, contrasts in NP balances and soil/groundwater NP supplies/contamination between Western and Eastern Europe are evaluated, i.e over fertilization and environmental problems in the Western part, and under fertilization and related agronomic and social problems in the Eastern part.

In this paper, some suggestions are made for making EU agro-environmental protection legislation much more effective. These suggestions are already built into the new Hungarian cost-saving and environmentally friendly RISSAC-RIA fertiliser recommendation system, based on the correlations found in the database of the Hungarian long-term field trials with N-, P-, and K-fertilisation (CSATHÓ *et al*., 1998, 2009). This fertiliser recommendation system and software was given the Innovation Grand Prize for Hungary in 2007, the highest national award in the field of innovation (VÁRALLYAY, 2008).

## Materials and methods

For calculating environmental NP balances, the OECD methodology was used (OECD, 1997). For evaluating the soil phosphorus status of the Western European countries, the review of STEÉN (1997), of the Central and Eastern European countries, the review of CSATHÓ *et al*. (2007) was used as the basis. For strengthening the reliability of the N balance calculations in the EU countries, ground water status of the same countries were used. For strengthening the reliability of the P balance calculations in the EU countries, in the other hand, soil P status of the same countries were taken into account.

## Results and discussion

### *Correlations between P-supplying capacity and P balance*

One fundamental characteristic of fertiliser recommendations aimed at environmental sustainability is (or should be) that on areas poorly supplied with a given nutrient, a quantity larger than that taken up by the crop is applied, slightly more than crop uptake on soil with moderate supplies, an amount equal to or slightly less than crop requirements on soils with good supplies, little or none on soils with very good supplies, and no P (K) fertiliser on soils with an excessive supply level (Fig. 1).

The P recommendations, normally shown as kg P $ha^{-1}$ $year^{-1}$, are usually based on soil test phosphorus (STP) values and the following is an example of a typical fertiliser strategy (Tunney *et al*., 2003): i) no P fertiliser required for optimum production for a number of years when STP is high, ii) maintenance P (replacing removals) required when STP is moderate, iii) build-up of P recommended when STP is low. An example of an approach used in Germany to recommend fertiliser application at field level is illustrated in Fig. 3. (VETTER and FRUCHTENICHT (1974). It is based on STP level and shows, for example, that at fertility class A (very low fertility), twice the maintenance P fertiliser dressing is recommended and at fertility class D, only 0.5 times the maintenance level is recommended. At fertility class E (very high fertility) fertiliser P is not recommended (Figure 1).

If this logic is followed in the 27 EU countries of Western Europe where the soils were far better supplied with phosphorus in the early 90s, far lower rates of (N) P should be applied and far lower (N) P balances are justified both from the agronomic and environment protection points of view than in the countries of Central and Eastern Europe, where P supplies were far poorer in the early 90s.

| Fertility Class | Fertiliser Ratio |
|---|---|
| E: Very high | 0 |
| D: High | 0.5 |
| C: Moderate | 1.0 |
| B: Low | 1,5 |
| A: Very low | 2.0 |

C = Maintenance

*Figure 1.* Phosphorus fertiliser recommendation for fields in Germany based on soil fertility class soil test phosphorus (STP) based on VETTER and FRUCHTENICHT (1974), cit: TUNNEY *et al*. (1997).

Let us see how far this theory is put into practice. Fig. 2 illustrates the correlation between the P supply index, indicative of the P status of the soil and the P balance.

In order to calculate the P supply index, a value of 1 was applied for areas very poorly supplied with phosphorus, 2 for poorly supplied areas, 3 for moderately well supplied areas, 4 for well supplied areas and 5 for very well supplied areas. This was then multiplied by the % of land belonging to the given supply category, i.e. by 0.1 for 10% of the land, by 0.2 for 20%, etc. The figures obtained for each category were then added to give the P supply index of the country.

A country very poorly supplied with phosphorus over 100% of its area would thus have a P supply index of 1.0, while the other extreme would be a country with very good supplies over 100% of its area, having a P supply index of 5.0. The introduction of a 6$^{th}$ category for excessive supplies of P would also be justified, but the necessary data is not available at present (CSATHÓ *et al*., 2009).

If P fertilisation were carried out in a manner acceptable from the agronomic and environment protection points of view, a negative correlation would have been plotted in Fig. 2., with P balances declining as the P supplies improved. By contrast, the opposite was observed in Europe in the early 1990s: the P balances in Central and Eastern Europe, where the P supply index was lowest, were the smallest and were in some cases negative (between -5 and -10 kg ha$^{-1}$ P), while Western European countries, which had the highest P supply indexes, had the most positive P balances, with surpluses of 18–40 kg ha$^{-1}$ P each year. This unfavourable situation (i.e. the polarization between the Western and Eastern part of the EU) has even accelerated and has become much worse since the introduction of the Nitrates Directive, as is clear from the cumulative nitrogen and phosphorus balance of European countries over the last 15 years.

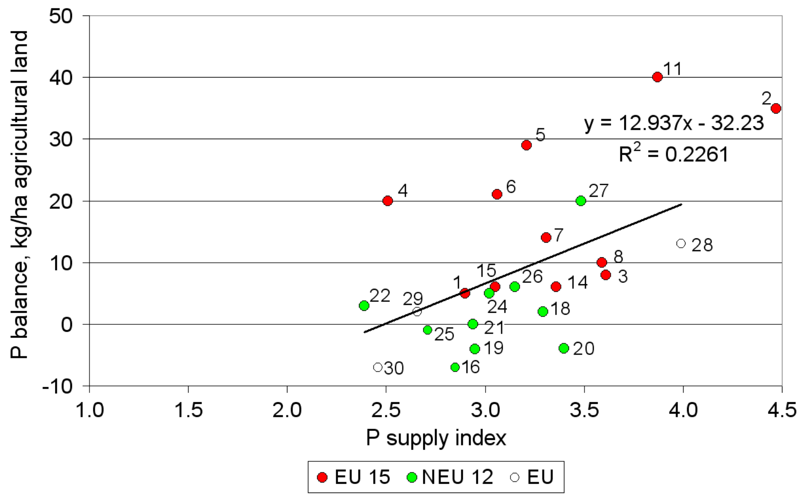


*Figure. 2.* Correlation between the soil P supply index and the P balance in the countries of Europe in the early 1990s (Numbers 1-30: see in text at Fig. 1)

***Cumulative N and P balances in the European Union***

The cumulative N balance of certain European countries, many of them EU member countries, are presented in Fig. 3. for the period 1991 to 2005. The Netherlands and Belgium lead the field for N balances. During the 15 years that have elapsed since the Nitrates Directive was introduced the total N surplus was 2800 kgha$^{-1}$ in Belgium and 3500 kgha$^{-1}$ in the Netherlands, and was also well above 2000 kgha$^{-1}$ in Denmark. It seems that the Nitrates Directive has not proven to be effective in reducing agricultural N(P) loads into the environment in the EU countries with the highest livestock densities.

The cumulative N balance was also above average in Germany, Norway and Ireland, while the countries of Central and Eastern Europe came last, as expected (Fig. 3).

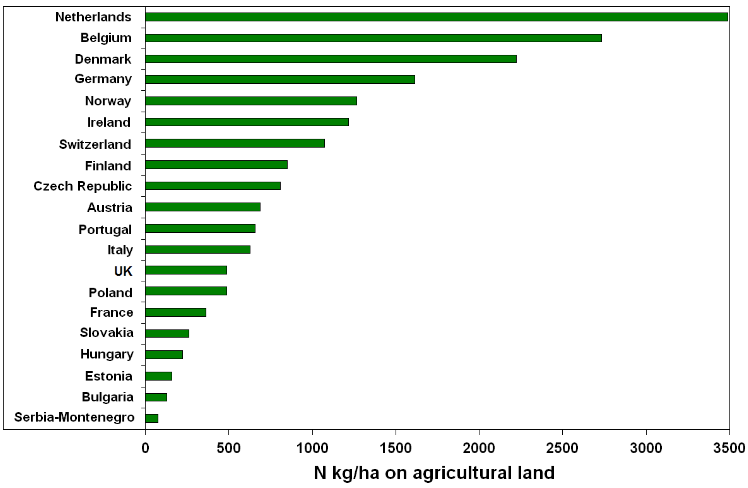


*Figure. 3.* Estimated cumulative N balance of European countries, 1991–2005 (CSATHÓ and RADIMSZKY, 2007, 2009)

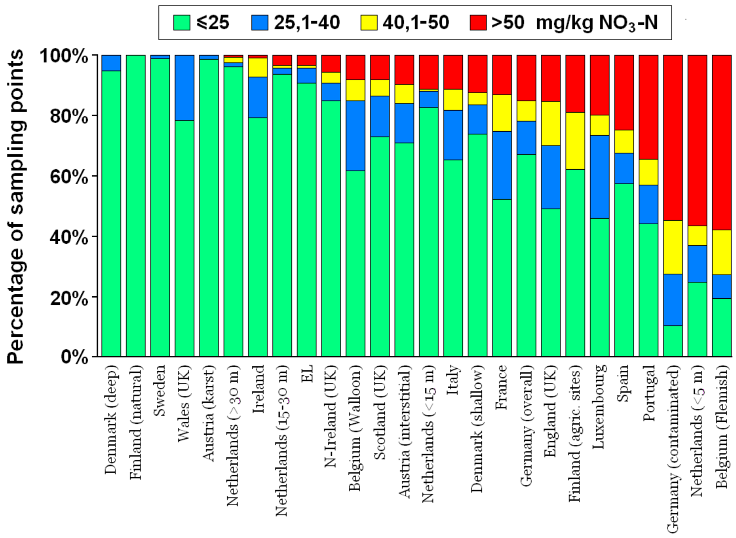


*Figure 4.* Ground water status of some EU 15 countries in the mid 2000's (HAMELL, 2007)

High correlation between the magnitude of cumulative N balances, as well as the measure of nitrate contamination of the groundwater can be registered in the EU15 countries (Fig.4)

The cumulated P balance is estimated for the period of 1991 to 2005 in Fig. 5. (CSATHÓ and RADIMSZKY, 2007, 2009). For countries where data was only available until 2002 or 2003, NP balance for the missing years was taken as being equal to the last recorded year.

The P surplus accumulated over this 15-year period was more than 400 kg ha$^{-1}$ P in the Netherlands and 300 kg ha$^{-1}$ P in Belgium (Fig. 5).

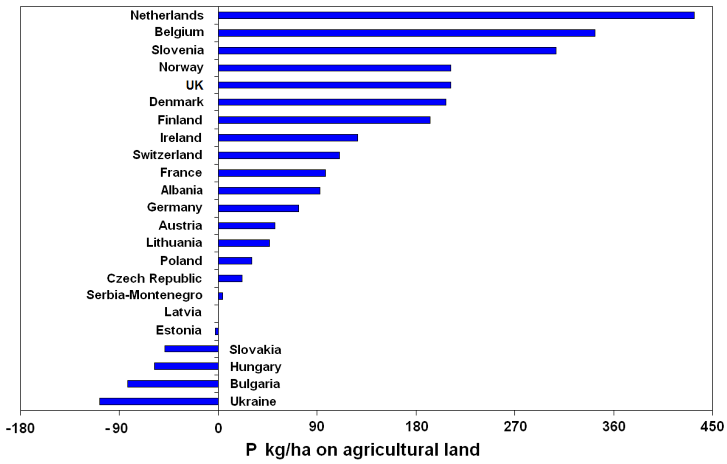


*Figure. 5.* Estimated cumulative P balance of European countries, 1991–2005 (CSATHÓ and RADIMSZKY, 2007, 2009)

Not to mention, due to the fact, that the most positive cumulative P balance was found in the countries with the highest soil P supplies, the situation has become even more threatening since 1991, indicating the inefficiency of the Nitrates Directive, implemented the same year, i.e. in 1991, aiming to regulate both N and P regime in the EU countries. Slovenia, Norway, Denmark and Finland also registered above-average increases in P over the last 15 years, and the Central and Eastern European countries were again at the bottom of the list.

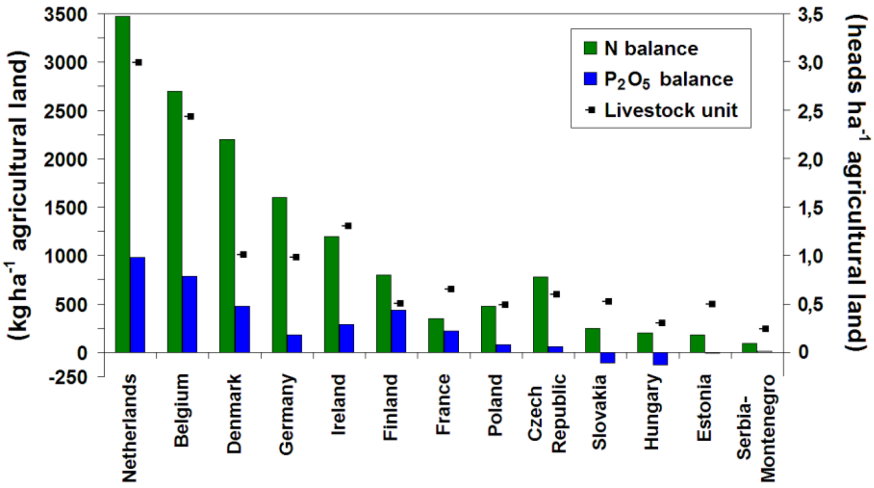


*Figure 6.* Estimated cumulative NP balance between 1991 and 2005 as well as livestock density of several European countries. (CSATHÓ & RADIMSZKY, 2009, 2011; CSATHÓ, 2010).

Within the EU countries, there is a strong positive correlation between cumulative NP balance and livestock densities (Fig. 6). It’s an important figure from the point of view of finding the solution to this situation.

*Table 1.* Ratios of the P loads to surface waters caused by various sectors in countries making up the watershed of the Danube in 1991, 1000 tonnes P, or %
(IJJAS and BÖGI, 1994; VOLLENBROEK, 1994, NÉMETH *et al.*, 1994)

| Country | Area within the Danube watershed 1000 km² | Danube watershed as a % of the total area | Population (1000 t P) | Population (%) | Agriculture: Point source (1000 t P) | Point source (%) | Agriculture: Diffuse (1000 t P) | Diffuse (%) | Agriculture: Total (1000 t P) | Total (%) | Industrial+atmospheric +background (1000 t P) | Industrial+atmospheric +background (%) | Total (1000 t P) | Total (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Germany | 59.6 | 16.7 | 2.0 | 33 | (-) | (-) | (2.0) | (33) | 2.0 | 33 | 2.0 | 34 | 6.0 | 100 |
| Austria | 80.7 | 96.2 | 4.0 | 66 | (-) | (-) | (1.8) | (29) | 1.8 | 29 | 0.3 | 5 | 6.1 | 100 |
| Czech Republic | 22.5 | 28.5 | 0.8 | 32 | (-) | (-) | (0.6) | (24) | 0.6 | 24 | 1.1 | 42 | 2.5 | 100 |
| Slovakia | 48.7 | 99.3 | 3.6 | 77 | (-) | (-) | (0.4) | (9) | 0.4 | 9 | 0.7 | 14 | 4.7 | 100 |
| Hungary | 93.0 | 100.0 | 8.1 | 75 | (-) | (-) | (1.1) | (10) | 1.1 | 10 | 1.6 | 15 | 10.8 | 100 |
| Slovenia | 15.2 | 75.0 | 1.0 | 21 | (0.2) | (4) | (1.9) | (40) | 2.1 | 44 | 1.7 | 35 | 4.8 | 100 |
| Croatia | 33.8 | 59.7 | 0.8 | 73 | (-) | (-) | (0.1) | (9) | 0.1 | 9 | 0.2 | 18 | 1.1 | 100 |
| Romania | 233.2 | 98.0 | 5.0 | 9 | (23.0) | (44) | (15.0) | (28) | 37.9 | 72 | 10.0 | 19 | 52.9 | 100 |
| Bulgaria | 48.2 | 43.4 | 1.6 | 57 | (0.5) | (18) | (0.5) | (18) | 1.0 | 36 | 0.2 | 7 | 2.8 | 100 |
| Total evaluated | 634.9 | 58.3 | 26.9 | 30 | (23.6) | (26) | (23.4) | (26) | 47.0 | 52 | 16.8 | 18 | 91.6 | 100 |

P loads to surface water from various sectors

Other countries in the Danube watershed, which were not evaluated: Moldavia: 8800 km$^2$; Ukraine: 36,300 km$^2$; Serbia-Montenegro + Bosnia-Herzegovina: 134,200 km$^2$; other: 2800 m$^2$; Total unevaluated area: 182,100 km$^2$; Total area of the Danube watershed: 817,000 km$^2$.

## Discussion

In a perfectly correct and justifiable manner, the European Union made investments in environment protection a strict condition for the accession of the Central European countries to the EU. One essential obligation was the satisfactory disposal of sewage, as a water protection measure. The necessity for this decision was underlined by the results of analyses carried out with PHARE funding in the framework of the Integrated Danube Research Program, which estimated the proportion of surface water (N) P loads caused by various sectors (Table 1.) (IJJAS and BÖGI, 1994**;** VOLLENBROEK, 1994**,** NÉMETH *et al*., 1994)**.** Due to the introduction of untreated sewage directly into surface waters, the NP load contributed by population waste was outstandingly high in Central Europe in the early 90s. The steps taken by the EU to protect surface waters have thus led to a dramatic reduction in point-source pollution caused by the (N) P contained in sewage.

The EU should be just as consistently strict in curbing the massive diffuse NP pollution caused by agriculture (Fig.7.).

| **Fertility Class** | **Fertiliser Ratio** |
|---|---|
| E: Very high | 0 |
| D: High | 0.5 |
| C: Moderate | 1.0 |
| B: Low | 1,5 |
| A: Very low | 2.0 |

A) Severe environmental threats (Western European EU countries)

B) Severe agronomic, social, and rural development problems (Eastern and Central European EU countries)

*Figure 7.* NP turnmover disorder in the EU27 countries, and its consequences (CSATHÓ, 2010)

This is the realistic figure of the NP fertilizer practice of the EU countries in the last 15 to 20 years and the consequences are severe environmental threats in some of the Western European EU countries or NUTS-2 regions and severe agronomic, social, and rural development problems in the rural areas of most Eastern European EU countries.

The European Union must decide whether it is willing to sacrifice environment protection, the handling of social problems and the interests of rural development for the sake of liberalizing agricultural markets. In some respects, Europe lags behind the United States in terms of agricultural environment protection. In many states effective legislation has been passed to reduce P loads of agricultural origin, despite the fact that the situation is far less serious than in many European countries (SHARPLEY *et al*., 1994; GARTLEY and SIMS, 1994). The directives passed by the EU should also be compulsory, not simply recommended.

So that to fit EU NP turnover to the description of „Union",

- Principles of the environmentally friendly / sustainable N fertiliser advisory system should be built in the EU Nitrates Directive urgently.
- A new, independent Phosphates Directive should be elaborated and implemented urgently.
- Principles of the environmentally friendly / sustainable P fertiliser advisory system should be built in the new EU Phosphates Directive.

**Conclusions**

In the opinion of the authors, without drastic measures, i.e., decreasing the livestock densities in the Benelux countries, in Denmark, Germany, in the Bretagne peninsula and in the Po valley, etc. to 75 LU/ 100 ha agricultural land, and without increasing livestock density to the same level in the new 12 EU countries, the threatening environmental problems in some of the Western EU countries, and the threatening economic, agronomic, social and rural development problems of the Central and Eastern European EU countries cannot be solved. In addition to this, all the advantages derived from these changes should go to the local farmers, farmers' associations and local communities, prior to lifting the moratorium on foreigner companies or private individuals ~~on~~ purchasing land and other agricultural properties in the Central and Eastern European new EU12 countries.

Due to possible climate changes, the threat of accelerated agricultural NP loads in the environment is increasing. To decrease this threat, the principles of economic and environmentally friendly plant nutrition practice must be urgently implemented amongst the farmers of the EU.

**Acknowledgement**

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This paper has been dedicated to the dialogue between the Western and the Eastern part of the EU, for the benefit of the common people in both macroregions.

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# MITIGATION OPTIONS FOR REDUCING NITROGEN LOSSES TO WATER FROM GRAZED DAIRY PASTURES IN SOUTHERN NEW ZEALAND

*Ross MONAGHAN*

AgResearch, Invermay, Private Bag 50034 Mosgiel, New Zealand

**Abstract**

There are a number of mitigation practices currently available that can reduce the impacts of intensive farming on water quality. However, each mitigation measure differs in its effectiveness, cost and likely impact on those waters, depending on factors that include soil type, climate, topography and the regional sensitivity of water bodies. Consequently, it can be difficult for land managers to select a mitigation measure or combination of mitigation practices which are most appropriate to their farm. To address this problem, a Toolbox of Best Management Practices (BMPs) has been assembled with a suite of options, an assessment of their cost and effectiveness, and an indicative ranking of where expenditure should be prioritised to ensure that maximum benefit is obtained for each dollar invested. Here we describe how this tool has been used to guide farm management decisions aimed at reducing N losses to water from dairy farms in southern New Zealand. Toolbox cost-effectiveness rankings show that substituting cow diet with low N feed, stream fencing and the provision of effluent storage are measures that deliver greatest benefit at least cost, although are estimated to collectively only reduce farm N losses by about 20%. Management strategies that target urine N deposited to fields in autumn or winter appear to be the next most cost-effective options for reducing N losses to water from case study farms. Such practices include applying the nitrification inhibitor dicyandiamide (DCD) to pastures, off-paddock wintering of cows, or restricting pasture grazing times during autumn. Wetlands are another cost-effective option for attenuating N in land drainage, although their efficiency depends on the degree to which drainage flows can be intercepted. In contrast, management changes that involve changing land use to less intensive dry stock farming systems or reducing dairy cow stocking rates are strategies that incur large opportunity costs and thus have relatively low cost-effectiveness.

**Keywords:** nitrogen losses, water quality, mitigation, dairy farming, grazing.

## Introduction

Deteriorating trends in water quality are of concern to many countries, particularly those of the industrialised West where intensive farming practices are often used to support high levels of food productivity. Most modern farming systems continually seek to improve farm business productivity through increasing outputs of saleable product or through decreasing input costs, or both. An inevitable outcome of the former strategy is increased land use intensity and farm inputs of feed, fertiliser and energy. Inefficient nutrient cycling and increased losses of nutrients to the environment are usually an unfortunate consequence of this pathway to economic productivity. Greater attention is now being focused on some of the off-site impacts of farming activities, particularly the consequences of nutrient enrichment of ground and surface waters. Intensive agriculture is known to emit significant amounts of nutrients, particularly nitrogen (N) and phosphorus (P) (GILLINGHAM and THORROLD, 2000; WATSON and FOY 2001; MONAGHAN *et al*., 2007). While these emissions are typically not large by agronomic standards, the transfer of pollutants from land to water can significantly impair water quality. These transfers have been shown to increase as farm inputs increase and systems intensify (e.g. LEDGARD *et al*., 1999; SCHOLEFIELD *et al*., 1993; WATSON *et al*., 2000). Livestock production systems are identified as an important sector that has contributed to water quality impairment due to such intensification. Potential solutions that can effectively address these negative consequences at reasonable cost are urgently needed (FAO 2006).

Recent and on-going research has shown that there are a number of technological measures or management practices that can potentially reduce N losses from farms to water. However, this research has also shown that it is important that mitigation measures are matched to the physical resources and management systems of individual farm businesses to ensure losses are reduced with maximum cost-effectiveness. Although conceptually a simple process, the actual practise of matching specific mitigation measures to individual farms is rather complex and requires a good understanding of N flows and losses and a sound knowledge of the capital and operational costs that may be incurred. Factors such as soil type, topography, existing farm infrastructure and lifestyle combine to influence farmer decisions about which measure is most appropriate for their situation. Farmers have also identified that they prefer to consider a suite of mitigation options so that they can match individual practices to their farm context, rather than have prescriptive practices imposed upon them. To assist with this decision-making process, a range of BMPs have been incorporated into a Toolbox of practices which documents the cost-effectiveness of each, thus providing an indicative ranking of where expenditure should be prioritised to ensure that maximum benefit is obtained for each dollar invested. Here we describe how this tool has been used to guide farm management decisions aimed at reducing N losses to water from dairy farms in southern New Zealand. These farms are characterised by their reliance on grazed pasture during the milking season from early spring until late autumn as the main source of animal feed, and the use of grazed brassica crops as the main source of winter forage.

## Approach

### *Mitigation measures contained within the Toolbox*

For the purposes of this assessment, a "typical" case study dairy farm was defined and then used to explore the most cost-effective options that were potentially available for reducing N losses. This case study farm was essentially based upon that described as a representative farm located in the Bog Burn catchment of southern New Zealand (MONAGHAN *et al*., 2008), but updated to reflect farm production and management characteristics based upon detailed farm survey interviews completed in 2009. The average farm size was 210 ha, with a stocking rate of 2.4 cows $ha^{-1}$ producing 14,000 litres of milk $ha^{-1}$. This was supported through the modest use of fertiliser N and P (100 and 37 kg $ha^{-1}year^{-1}$, respectively), clover N fixation (estimated at c. 80 kg $ha^{-1}year^{-1}$) and the annual importing of 700 kg DM per cow as feed supplements.

The suite of N mitigation measures currently available for mitigating N losses from dairy farms in southern NZ are listed in Tables 1 and 2. Table 1 lists those measures that involve farm system management changes, which may sometimes involve relatively complex adjustments to management operations e.g. the installation of a cow housing facility. Edge of field measures for intercepting N losses or protecting riparian margins are documented in Table 2. This Toolbox of measures has been assembled to potentially reduce both nitrogen (N) and phosphorus (P) losses, but for the purposes of this paper we confine our analysis to measures for reducing N losses to water. For each measure, the Toolbox provides a set of three metrics:

### *1. BMP effectiveness*

The effectiveness of any single measure for reducing farm-scale losses of N to water is determined using the OVERSEER® Nutrient Budgeting program, hereafter referred to as *Overseer* (WHEELER *et al*., 2003; software freely available and can be downloaded from www.overseer.org.nz). This modelling tool allows users to examine nutrient flows within a farm, and potential environmental impacts, under a range of user-defined scenarios that can be constructed. The advantage of using such a modelling tool is that it can account for both "resource" (e.g. soil type, drainage, rainfall, topography and regional location) and "management" risk factors (such as farm type (dairy, sheep, beef or deer), feed type, effluent management practices, soil Olsen P level, irrigation use and extent of stream fencing) that are known to influence the amount of N lost from farms to water. It has the additional advantage that it is now widely used by dairy farmers and their advisors and thus provides some consistency between farms and farm scenarios constructed by these rural professionals. For most of the Toolbox measures that were applied to the case study farm under consideration here, it was assumed that each was applied singly. The exception was the restricted autumn-winter grazing system, where it was assumed that full off-wintering would take place before implementing a restricted autumn grazing regime. An estimate of the range in the effectiveness of each mitigation measure is provided, which accounts for some of the management uncertainty and variability associated with implementing each mitigation option. The regional variability in the effectiveness of some of the mitigation options was accounted for by the in-built functions within the Overseer tool (e.g. responses to nitrification inhibitors applied to pasture).

### *2. BMP cost*

The second metric provided by the Toolbox is an estimate of the annualised net cost of implementing each mitigation measure. For simplicity, the current Toolbox assumes a number of default values for cost components, such as the opportunity cost of capital (8%), depreciation, maintenance, additional labour and feed, and revenue foregone as a result of land lost to production. Any financial benefits expected from implementing measures are deducted from the net overall annualised cost. These benefits can be particularly important where a measure increases productivity (e.g. extra pasture growth from the use of nitrification inhibitors) or reduces farm operational costs (such as avoiding off-farm cow wintering fees if the animals are wintered under a Herd Shelter on the home farm). Some of the key costing assumptions made are listed in Tables 1 and 2. These costs can vary considerably between farms, depending on factors such as farm management system, existing farm infrastructure, soil type, topography, climate and farmer lifestyle preferences. Future upgrades to the Toolbox functionality will therefore allow users to enter their own cost assumptions if they prefer, instead of using the default values provided in the tool.

### *3. BMP cost-effectiveness*

The cost-effectiveness of each mitigation measure is computed by dividing the annualised net cost of each option by the quantity of N that is estimated to be conserved due to the implementation of the mitigation measure. This metric can be used to rank the mitigation measures in order of where best value for money is likely. A negative value for cost-effectiveness implies that the measure will actually result in a net benefit to the farmer whilst reducing farm N losses to water. Conversely, large positive values indicate that the measure will incur a large net cost for each kilogram of N conserved.

*Table 1*. Farm system management options available for reducing N losses to water from a case study dairy farm in southern NZ. Costing assumptions are reported in NZ dollars.

| Mitigation option | Cost assumptions |
|---|---|
| Nitrification inhibitors | 1. 2 DCD applications per year @ \$65 per application.<br>2. Pasture response to DCD = 2% |
| Off-paddock cow wintering | 1. Wintering costs increase from \$220 to \$370 cow$^{-1}$ due to extra capital (housing) costs<br>2. Use of winter pad not assumed to intensify farming |
| Restricted autumn grazing (with off-wintering) | 1. Winter cost components as per winter pad option above.<br>2. Autumn pasture grazing times restricted to 4 hours per grazing; animals housed under winter pad when |
| Use of low N feed | 1. Barley grain used to replace the importation of 2 T pasture silage DM ha$^{-1}$.<br>2. Net annualised cost calculated as a function of milk price (\$0.58 L$^{-1}$) and cost of imported feed (\$340 T |
| Nil N fertiliser | 1. Change in profitability calculated as a function of milk price and cost fertiliser N (further details available in Monaghan *et al*., 2008). |
| Deferred effluent (slurry) irrigation | 1. Additional pond storage requirement of 3.5 m$^3$ cow$^{-1}$, incurring a capital cost of \$40 cow$^{-1}$.<br>2. Opportunity cost of capital = 8%<br>3. Additional operational costs off-set by pasture yield responses to effluent. |
| Land use change to sheep farming | 1. Estimated net annualised cost for case study farm of \$1500 ha$^{-1}$ represents the opportunity cost of lost profit between dairy and sheep farming. |

*Table 2.* Edge-of-field mitigation measures available for reducing N losses to water from a case study dairy farm in southern NZ. Costing assumptions are reported in NZ dollars

| Mitigation option | Cost assumptions |
|---|---|
| Constructed wetlands | 1. Occupy 1% of farmed area<br>2. Setup cost of \$800 $ha^{-1}$. |
| Facilitated natural wetlands | 1. Occupy 3% of farmed area<br>2. Planting and fencing costs of \$150 $ha^{-1}$ for each. |
| Stream fencing | 1. Assumes 1 m of productive land is lost for each m of fencing still required.<br>2. 2-wire electric fence required, costing \$3 $m^{-1}$.<br>3. Stream density of 20 m $ha^{-1}$ assumed, with 20% remaining to be fenced.<br>4. Riparian margin planting costs of \$4 $m^{-2}$. |
| Grass buffer strips (4 m width) | 1. Assumes 20% of the grass buffer area was non-productive.<br>2. For a stream density of 20 m $ha^{-1}$, profit loss due to loss of productive land = \$30 $ha^{-1}$.<br>3. 2-wire electric fence required to protect buffer areas, costing \$122 $ha^{-1}$.<br>4. Annual maintenance cost for weed and vegetation control of \$10 $ha^{-1}$. |

**Results and discussion**

The cost of any mitigation measure is usually one of the most important criteria influencing decisions made by land managers who are considering options for reducing the impacts of their farm on water quality. If cost was the sole criteria for selecting a measure from the BMPToolbox, then substituting a low N feed, such as barley grain, for the 700 kg pasture silage DM that is imported onto the farm and fed per cow would be the preferred option for our case study farm, and would actually deliver a slight increase in profitablility (Table 3). The next least-costly mitigation measures are the provision of effluent (slurry) storage, fencing the remaining lengths of stream to prevent stock access and installing a 4 m grass buffer around farm streams. In contrast, by far the least profitable action for reducing N losses to water would be to convert the land back to sheep farming. The large cost of doing this reflects the very high opportunity cost of reduced profit from sheep compared to dairy farming. Eliminating N fertiliser inputs is another expensive option, again due to the opportunity cost of foregone profit, as are measures that require large capital investment, such as off-paddock cow wintering shelters.

Assessments of the effectiveness of each of the mitigation measures shown in Table 3 often show a contrasting pattern of ranking to that observed for cost. The 4 least-costly measures identified above (low N feed substitution, provision of effluent storage, stream fencing and grass buffer strips) are also the least effective, each delivering a 12% or less reduction in farm N losses to water. In contrast, the expensive options of land use change to sheep farming and off-paddock wintering (and restricted autumn grazing) using animal shelters are predicted to deliver reasonably large reductions in N losses.

Assessments of the cost-effectiveness of each of these mitigation measures provides a metric that can help to guide management decisions to identify the greatest benefit from each dollar invested. For our case study farms, a cost-effectiveness ranking shows that substituting the cow diet with low N feed, stream fencing and the provision of effluent storage are the measures that deliver greatest benefit at least cost (Figure 1).

*Table 3.* Toolbox estimates of the costs and effectiveness of a range of mitigation measures available for reducing N losses to water from a case study dairy farm in southern NZ. Costs are reported in NZ dollars

| Measure | Cost, \$ $ha^{-1}yr^{-1}$ | Effectiveness, % |
|---|---|---|
| **Farm management changes** | | |
| Low N feed substitution | -11[a] | 5 |
| Deferred effluent irrigation | 8 | 5 |
| Nitrification inhibitor | 70 | 19 |
| Nil N fertiliser | 437 | 19 |
| Off-paddock wintering | 167 | 27 |
| Restricted autumn-winter grazing | 188 | 60 |
| **Edge-of-field measures** | | |
| Grass buffer strips | 48 | 3 |
| Stream fencing | 11 | 12 |
| Facilitated natural wetlands | 104 | 17 |
| Constructed wetlands | 110 | 17 |
| | | |
| **Land use change** | 1492 | 55 |

[a]negative value indicates a net financial benefit

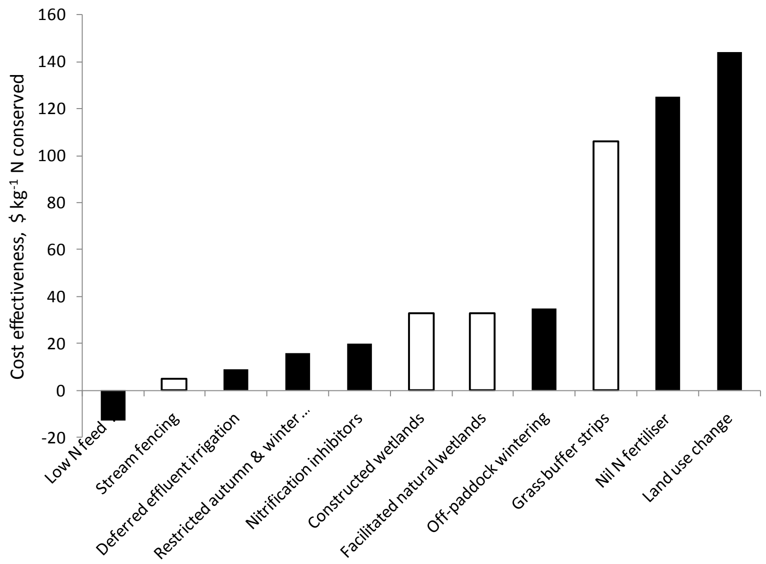


*Figure 1*. The estimated cost-effectiveness (\$ per kg of N conserved) of a range of measures for reducing N losses to water from grazed dairy pastures in southern New Zealand. Solid and empty bars denote farm system management changes and edge-of-field measures, respectively

Management strategies that target urine N deposited in autumn or winter appear to be next the most cost-effective options for reducing N losses. Such practices include applying the nitrification inhibitor dicyandiamide (DCD) to pastures (Figure 2), off-paddock wintering of cows, or restricting pasture grazing times during autumn. The latter option was modelled assuming that this management practise would only be implemented if a cow housing facility was avaiable, which in practise is relevant to those farms that already winter off-paddock. Wetlands are another cost-effective option for attenuating N in land drainage, although their efficiency depends on the degree to which drainage flows can be intercepted. In contrast, management strategies that involve changing land use to less intensive dry stock farming systems or reducing dairy cow stocking rates by reducing N fertiliser inputs are strategies that incur net costs of greater than NZ$100 per kilogram of N conserved.

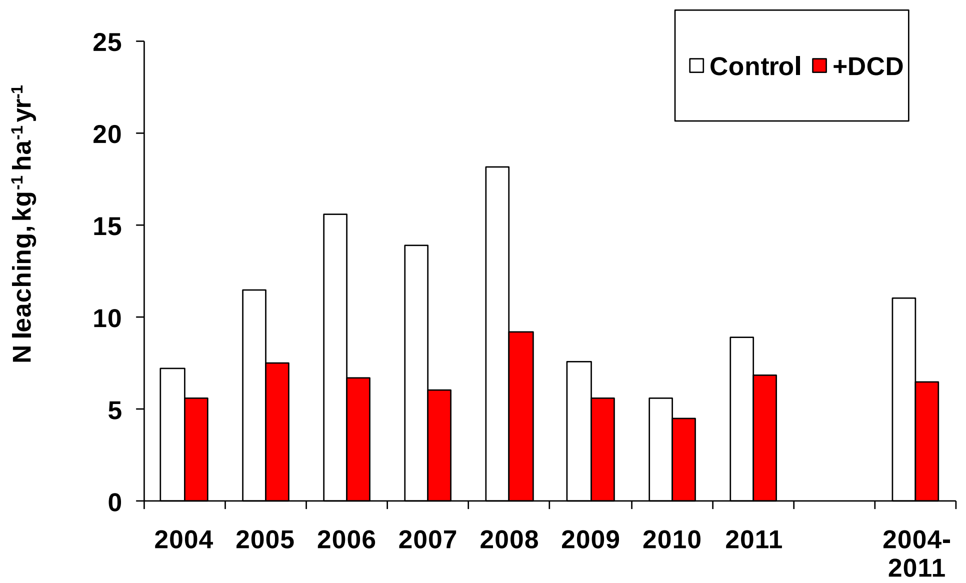


*Figure 2*. The long-term effectiveness of the nitrification inhibitor dicyandiamide (DCD) in reducing nitrate-N leaching losses from a mole-pipe drained pasture grazed by dairy cows in southern New Zealand. Applications of DCD in late autumn and early spring have been made to the experimental site since 2004; further experimental details can be found in MONAGHAN *et al*., 2009.

The Toolbox functionality described here provides a preliminary screening of some of the most cost-effective measures available for mitigating N or P loss from pastoral farms. Ideally, this screening would be the basis for more detailed scenario evaluations using expert farm system modelling tools such as FarmaxDairy (BRYANT *et al.,* 2010) to fully evaluate whole-system changes to the farm business. These detailed scenario evaluations will help to tailor mitigation practices to fit individual farms which vary widely, even within regions, in their management approaches and lifestyle expectations. In practise, the decision to take any course of mitigation action depends on an wide range of factors than those considered here. Although hard to quantify, the ancillary benefits of many measures are important aspects that may often even out-weigh the simple cost-benefit analysis undertaken within a Toolbox calculation. A relevant example is the decision to protect or ”faciliate” natural wetlands, which is often undertaken for purposes such as the enhancement of biodiversity values or for easier management of livestock movements. The Toolbox functionality described here is thus only a tool to help guide on-farm decision-making and provide some indication of cost and effectiveness. It is intended that additional mitigation practices will be added to the Toolbox as new research information becomes available.

### Conclusions

Consideration of the environmental impacts of farming practices such as grazing, effluent and riparian management is now an integral part of running a farm business. Although there are a number of mitigation measures available that can potentially reduce losses of N (and P) from farms to water, choosing the most cost-effective option has been a challenging task. The development of the BMPToolbox attempts to make this task easier by guiding users to measures that deliver greatest benefit at least cost. Case study analysis for a typical dairy farm in southern New Zealand suggest that there are a number of N mitigation measures that could be easily implemented to provide greatest benefit at least cost, although are estimated to collectively only reduce farm N losses by about 20%. More costly management strategies that target urine N deposited to fields in autumn or winter appear will be required to deliver larger reductions in farm N losses to water. The most expensive and least cost-effective measures are those that incur large opportunity costs in terms of foregone profit, such as changing land use to less intensive (and less profitable) dry stock farming systems, or reducing dairy cow stocking rates.

### Acknowledgements

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# MANAGING CATTLE SLURRY APPLICATION TIMINGS TO MITIGATE DIFFUSE WATER POLLUTION

*E. SAGOO[1] – J.P. NEWELL PRICE[2] – J.R. WILLIAMS[3] – R.A. HODGKINSON[3] – B.J. CHAMBERS[2]*

[1]ADAS Boxworth, Battlegate Road, Boxworth, Cambridge, CB3 8NN, UK, *e-mail:Lizzie.Sagoo@adas.co.uk*
[2]ADAS Gleadthorpe, Meden Vale, Mansfield, Notts. NG20 9PD, UK
[3]ADAS Boxworth, Battlegate Road, Boxworth, Cambridge, CB3 8NN, UK

**Abstract**

Around 47 million tonnes of livestock slurry supplying *c.*210,000 tonnes of nitrogen (N) and *c.*50,000 tonnes of phosphorus (P) are applied to agricultural land in the UK each year. Efficient utilisation of manure nutrients is essential to reduce diffuse water pollution. Moreover, organic manures are considered to be one of the main causes of controllable nutrient pollution in UK farming systems.

This paper summarises results from a drained clay soil study site in Oxfordshire (England) where the impact of different slurry application timings (autumn, winter and spring) on losses of agricultural pollutants to water (nitrate, ammonium and phosphorus) were quantified over three drainage seasons. The autumn slurry applications to arable land presented the greatest risk of nitrate-N loss to drainage waters ($P<0.05$), with losses in the range 8-11% of total N applied, compared with 2-6% of total N applied from the winter timings. However, ammonium and phosphorus (P) losses in drainage waters following the autumn slurry applications were low. In contrast, slurry applications in winter and spring resulted in elevated ammonium and P concentrations/losses in drainage waters, reflecting the rapid connectivity between the soil surface and field drains when slurry applications are made to 'wet' soils.

The results from this drained clay soil study site show that spring slurry application timings present the lowest risk and autumn timings the highest risk of nitrate leaching loss. However, slurry applications to 'wet' soils, particularly in winter, but also in spring, are likely to result in elevated ammonium and P concentrations in drainage waters (an example of 'pollution swapping'). In order to minimise the risks of diffuse water pollution, farmers will need to ensure that they have sufficient (over-winter) slurry storage capacity to provide the flexibility to spread slurry when soils are 'dry' in spring and summer (i.e. ideally when the soil moisture deficit is >20mm).

**Keywords:** Cattle slurry, diffuse pollution, nitrate, ammonium, phosphorus.

## Introduction

The UK Government is committed to improving water quality under the EU Water Framework Directive (WFD) and this can only be achieved through reducing pollutant losses from multiple sources. Nutrient emissions from agricultural land make a significant contribution to elevated nutrient concentrations in ground and surface waters in the UK and Europe (EEA, 2005). Estimates suggest that agricultural land contributes 60-70% of the nitrogen (N) and around 25% of the phosphorus (P) load to UK waters (HUNT *et al.*, 2004; DEFRA, 2007; WHITE and HAMMOND, 2007). Losses of P are up to an order of magnitude smaller than those of N, but are often more significant with respect to freshwater eutrophication (WITHERS and LORD, 2002). It is therefore important that emissions of nitrogen and phosphorus are reduced in order to improve water quality from headwaters to estuaries.

Reducing N and P concentrations in water is a major challenge, as there are multiple and widespread anthropogenic nutrient sources, such that virtually all aquatic systems are now enriched with nutrients to some extent. Nevertheless, significant improvements are possible and under the Nitrates Directive (CEC, 1991) actions have been taken to reduce N and P losses from agriculture, with the ultimate aim of reducing nitrate concentrations in all surface waters and groundwater to below 50 mg $l^{-1}$. These actions include autumn/winter closed periods for the spreading of fertilisers and high readily available N organic manures to reduce the overall size of the late autumn/winter soil nitrate pool that is available for over-winter loss through leaching (CHAMBERS *et al*., 1999; LORD *et al*., 2007). By contrast, for phosphorus (other than the WFD), there is no unified European legislative driver to reduce P losses to water, as the relationship between nutrient load, P concentration and trophic state is site specific depending on the type and sensitivity of the water body (EDWARDS *et al*., 2000). However, numerous methods can be employed to mitigate P losses from agriculture (e.g. KAY *et al*., 2009; NEWELL PRICE *et al*., 2011; SCHOUMANS *et al*., 2011).

Livestock manures are an important source of N and P and their careful management can have a large mitigating effect, particularly on local water quality. Indeed, the EU Commission recognises that the land application of livestock manures (particularly slurry) is one of the *main causes of controllable diffuse pollution* in present day farming systems (NICHOLSON *et al*., 2011). Around 47 million tonnes of livestock slurry supplying *c.*210,000 tonnes of nitrogen (N) and *c.*50,000 tonnes of phosphorus (P) are applied to agricultural land in the UK each year (WILLIAMS *et al.,* 2000; CHAMBERS *et al.,* 2000). The spreading of slurry is a 'high' risk activity, as it is a source of biologically active N and P, and the liquid component of slurry provides a medium for mobilisation and transport of pollutants. Slurry applications can result in 'incidental' losses of ammonium and P, via surface run-off and drain flow (HAYGARTH and JARVIS, 1999; SMITH and CHAMBERS, 1998; WITHERS *et al*., 2003). Also, they can result in nitrate losses to water following the conversion (nitrification) of ammonium to nitrate before or during the over-winter drainage period.

Applications of slurry therefore represent a potential source of elevated N and P concentrations in surface run-off and drainflow waters. Nutrient sources are commonly classified into point (e.g. sewage treatment works and direct discharges from septic tanks), diffuse (e.g. surface and drainflow run-off) and intermediate (e.g. septic tanks with soak aways or farmyard and road/track run-off). However, there are overlaps between these categories and losses from slurry spreading, although often regarded as diffuse, can result in 'elevated' water concentrations (similar to a point or intermediate source) for short periods following application (CHAMBERS and SMITH, 1998; WITHERS *et al*., 2003). It is therefore important that slurry spreading is optimised in terms of application rate, timing and method of application.

Most previous UK research investigating nutrient losses following slurry applications to agricultural soils has focused on sandy soils (SMITH *et al*., 1994; BECKWITH *et al*., 1998; SMITH and SHEPHERD, 2000). These soils (because of their permeability) are widely perceived as presenting the greatest risk of nitrate loss and because they commonly overlie aquifers, losses are of direct significance to potable groundwater quality. The processes controlling N and P losses from drained clay soils (JOHNES and HODGKINSON, 1998) are known to be markedly different from those operating in structureless sandy soils (LORD and SHEPHERD, 1993; WITHERS and LORD, 2002). Similarly, the pattern and amount of N and P losses from clay soils is likely to differ from sandy soils. On sandy soils, drainage occurs slowly over-winter by piston displacement in the unsaturated phase, with wetting fronts moving to depth at rates of a few metres a year depending on drainage volumes and the pore volume of the soil and base rock. However, on drained soils (which cover an estimated 7.5 million hectares of land in the UK) the rapid transfer of water from the soil surface to field drains, via soil macropores (i.e. 'by-pass flow'), could potentially lead to higher nutrient concentrations and losses following slurry application, with transit times influenced by rainfall volume and intensity (GOSS *et al*., 1983).

This paper reports the results of a field experiment set up to better understand the processes controlling nutrient losses from drained clay soils and how strategies to minimise the loss of one pollutant (e.g. nitrate to water) interact with losses of another (e.g. phosphorus and ammonium-N to water), so that 'win-win' and 'pollution swapping' situations could more clearly be identified. The paper seeks to test the hypothesis that moving slurry applications from autumn to spring is not only effective in reducing nitrate losses to water, but might also be effective in reducing ammonium and phosphorus losses.

## Materials and methods

### *Site*

The experiment was carried out at Brimstone Farm, Oxfordshire, UK (National Grid Reference SU 248 946). The experimental platform consisted of 18 hydrologically isolated plots (40 m x 48 m) on clay soil of the Denchworth Series (54% clay), which 'cracks' strongly during summer drying and is 'saturated' for long periods over the winter months. The site was on a 2% slope and was in continuous arable cropping for over 20 years before grass was established on 9 of the 18 plots in August 2001. The two land uses investigated were: i) arable and ii) 'arable reversion' grassland (Figure 1).

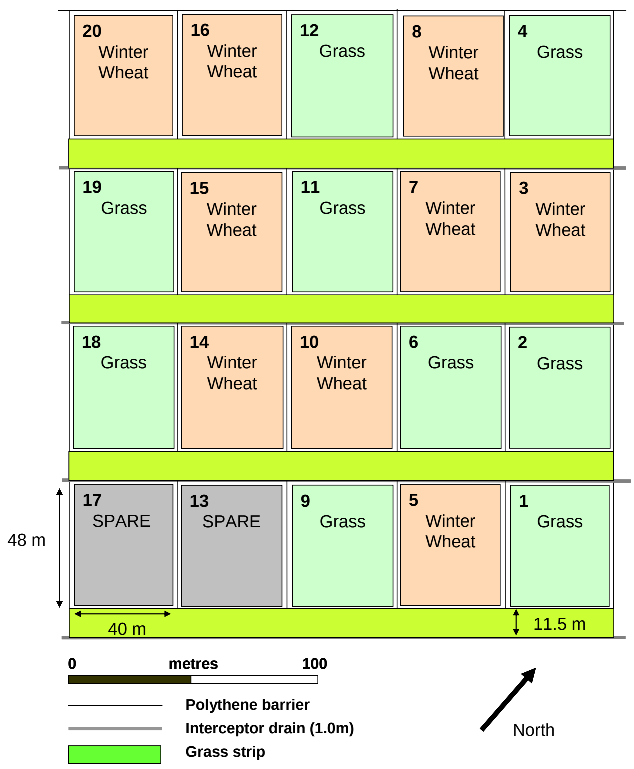


*Figure 1.* Plot layout at Brimstone Farm

Each plot had pipe drains at 48 m spacing and 90 cm depth, with permeable fill to within 30 cm of the surface. Secondary mole drainage was carried out at 50 cm depth and 2 m spacing, and was renewed periodically (as required) to maintain an effective drainage system and remove excess water from the crop rooting environment. Surface run-off was collected in *c.*10 cm deep gullies, along the downslope side of each plot. Each plot was hydrologically isolated by polythene barriers inserted to 1.1 m depth parallel to the 2% slope and by interceptor drains at 1.2 m depth in gravel filled trenches along the downslope plot margins.

Each plot was managed as though it was an individual field with all operations (i.e. cultivations, fertiliser, agrochemical and slurry applications) made using conventional farm machinery.

**Treatments – slurry applications**

In cropping seasons 2003-04, 2004-05 and 2005-06, cattle slurry was applied to the arable and grassland plots at three separate timings: autumn (August-October), winter (November-January) and spring (February-April). The target application rate was 45 $m^3$ $ha^{-1}$, with mean organic manure nutrient loading of *c*.120 kg total N $ha^{-1}$ and *c*.20 kg total P $ha^{-1}$. In summary, there were three treatments on arable land and grassland and three replicates of each treatment (18 plots in total), arranged in a randomised design.

Cattle slurry was applied using an 11 $m^3$ Joskin tanker, fitted with a 12 m trailing hose boom. For all application timings, cattle slurry was sourced from a commercial dairy farm adjacent to the site. On the grassland plots, all slurry applications were surface applied. On the arable plots, autumn slurry applications were made to cereal stubbles and ploughed down within 48 hours of application, with the winter and spring timings top-dressed onto the growing winter wheat crops.

The MANNER decision support system (*CHAMBERS et al*., 1999) was used to predict the crop available N supplied by each of the contrasting slurry application timings to the grassland and arable farming system plots. The N requirements of the grassland and winter wheat crops were based on guidance detailed in Defra's "Fertiliser Recommendation booklet - RB 209" (*ANON.*, 2000) and manufactured fertiliser N applications adjusted to take into account the N supplied by the contrasting slurry application timings. The grassland plots were cut for silage in late May/early June each year, followed by grazing with sheep in summer/autumn 2004 and 2005.

**Measurements**

Rainfall was measured using a tipping bucket rain gauge connected to a datalogging system. Drainflow volumes were measured continuously using v-notch weirs. Dataloggers were used to record flow data electronically and to control the automatic water samplers. Water samples were collected on a flow proportional basis and analysed for total N, nitrate-N ($NO_3$-N), ammonium-N ($NH_4$-N), total P (TP), total dissolved P (TDP), molybdate reactive P (MRP) and suspended sediment (*Anon.*, 1986). Note: Total N, MRP and suspended sediment data are not discussed in this paper.

The water samplers were programmed to collect drainage water samples on a flow proportional basis every 1 mm of drainflow for the first 25 mm of drainage after application, every 2 mm for the next 50 mm of drainage, every 4 mm for the next 125 mm of drainage and every 8 mm for subsequent drain flow volumes. Nutrient and sediment concentrations were combined with measured drainflow (and surface run-off) volumes to calculate total losses (kg $ha^{-1}$) over each drainage season from the contrasting treatments.

**Results**

***Drainage volumes***

Mean drainflow volumes from the arable plots were 130 mm in 2003/04, 125 mm in 2004/05 and 109 mm in 2005/06, compared with the long-term arable average of 204 mm (1978/79 to 1999/00). Mean drainflow from the grassland plots was 77 mm in 2003/04, 91 mm in 2004/05 and 55 mm in 2005/06. Over the three drainage seasons, the mean drainflow volume from the grassland plots was 74 mm, which was *c*.40% lower than from the arable plots (mean 121 mm), with drainflow from the grassland plots beginning 1-2 weeks later than from the arable plots. The lower drainage volume and later return to field capacity on the grassland plots (compared with the arable plots) reflected greater evapo-transpiration losses and soil moisture deficits due to grass growth in summer and early autumn.

### *Nitrate losses*

On the arable plots in all three years, nitrate-N ($NO_3$-N) concentrations in drainage waters were greatest following the autumn slurry application timings, peaking in the first 5-10 mm of drainage at *c.*130 mg $l^{-1}$ $NO_3$-N in 2003/04 and *c.*70 mg $l^{-1}$ $NO_3$-N in 2004/05 and 2005/06. On the plots where autumn slurry was not applied, peak $NO_3$-N concentrations at the start of drainage were 20-30 mg $l^{-1}$ lower than these values. Indeed, on the arable plots following autumn slurry applications, slurry N accounted for a mean of 41% of total nitrate-N leaching losses.

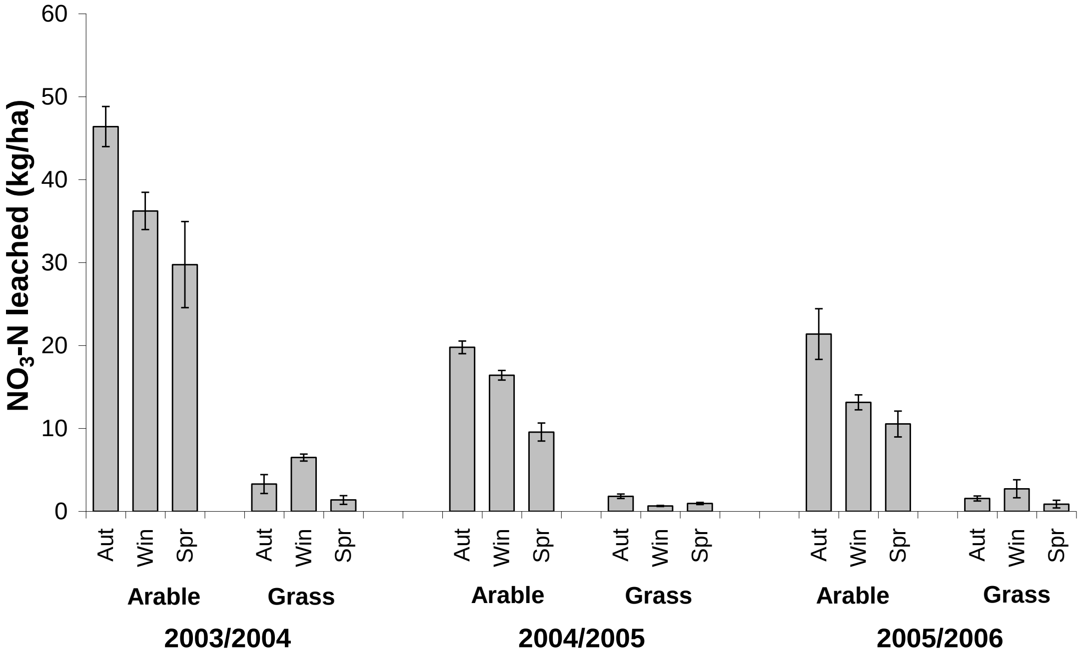


*Figure 2.* Nitrate-nitrogen losses in drainage waters (2003/04 to 2005/06)

The winter and spring slurry applications did not markedly increase $NO_3$-N concentrations in the drainage waters. For the winter applications, this was probably due to the cold and wet soil conditions delaying the nitrification of slurry $NH_4$-N to $NO_3$-N. Nitrate-N concentrations declined following the first 25-50 mm of drainage to *c.*40 mg $l^{-1}$ in 2003/04 and 10-20 mg $l^{-1}$ in 2004/05 and 2005/06.

On the arable plots, total nitrate-N leaching losses were greatest ($P$<0.05) following the autumn slurry application timings in all 3 years (Figure 2). Also, nitrate-N losses following the winter slurry application timings were (numerically) greater than from the spring treatments, although these differences were only significant ($P$<0.05) in 2004/05. Nitrate-N losses following the autumn and winter slurry application timings (up to the time fertiliser N was applied in the spring) were equivalent to 11% and 5% of total slurry N applied in 2003/04, 10% and 6% of total slurry N applied in 2004/05, and 8% and 2% of total slurry N applied in 2005/06, respectively.

On the arable reversion grassland plots, nitrate-N concentrations in drainage waters were significantly lower ($P$<0.05) than from the arable plots in all three study years. Mean $NO_3$-N concentrations, until manufactured fertiliser N was applied in the spring, were *c.*6, 1 and 3 mg $l^{-1}$ N in 2003/04, 2004/05 and 2005/06, respectively. Notably, on the grassland plots there was no effect (*P*>0.05) of slurry application timing on drainage water $NO_3$-N concentrations in all 3 years. In other words, neither the autumn nor the winter slurry applications on the grassland plots resulted in a significant increase in $NO_3$-N concentration in drainage waters. Similarly, on the arable reversion grassland plots slurry application timing had no effect (*P*>0.05) on $NO_3$-N leaching losses, with mean losses of between 1 and 6 kg N $ha^{-1}$.

In summary, slurry application timing had an effect on nitrate leaching losses from the arable plots, with the highest losses (*P*<0.05) following the autumn slurry applications, while on the arable reversion grassland plots slurry application had no effect (*P*>0.05) on nitrate leaching losses.

### *Ammonium-N losses*

Background drainage water ammonium-N concentrations were generally below the limit set in the Freshwater Fish Directive (FFD) of 0.78 mg $l^{-1}$ $NH_4$-N (Figure 3). On the arable plots, the autumn slurry application timings had no effect on $NH_4$-N concentrations in drainage waters in all three study years. On the grassland plots, autumn slurry application timings had no effect on $NH_4$-N concentrations in drainage waters in two of the three study years; the exception was in autumn 2004 where concentrations peaked at *c.*1 mg $l^{-1}$ $NH_4$-N. In contrast, the winter slurry applications in 2004/05 and 2005/06 resulted in elevated (i.e. >0.78 mg $l^{-1}$ $NH_4$-N) concentrations in drainage waters from both the grassland and arable plots. Similarly, following the spring slurry applications in 2004 and 2005 to both the arable and grassland plots, and in spring 2006 to the arable plots, drainage water $NH_4$-N concentrations were elevated.

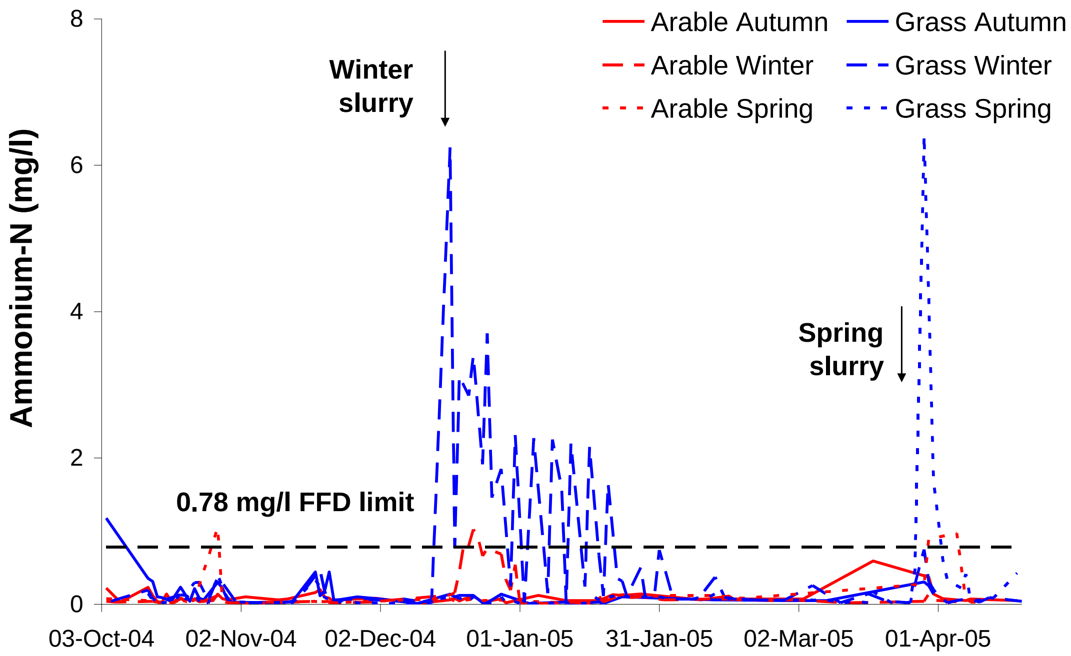


*Figure 3.* Ammonium-N concentrations in drainage waters (2004/05)

In 2003/04, the dry autumn meant that when the winter slurry treatment was applied on 17 December the soils had not fully 'wetted-up' (cumulative drainflow up to the slurry application was 16 mm on the arable plots and 1 mm on the grassland plots) and there was no effect of slurry application on $NH_4$-N concentrations in drainage waters from both the grassland and arable plots. In contrast, where slurry was applied on 3 March to soils that had fully 'wetted-up' and *c.*20 mm of rain fell 9-12 days after the slurry was applied, $NH_4$-N concentrations in drainage waters peaked at 3.9 and 4.5 mg $l^{-1}$ from the arable and grassland plots, respectively.

In 2004/05, heavy rainfall soon after both the 16 December (21 mm 1-2 days following application) and 23 March (30 mm 6-7 days after application) slurry applications to soils that had fully 'wetted-up' resulted in peak $NH_4$-N concentrations of 6.3 mg $l^{-1}$ from the grassland plots (*c*.8-fold greater than the EC FFD limit of 0.78 mg $l^{-1}$) and 1 mg $l^{-1}$ from the arable plots, respectively (Figure 3).

In 2005/06, heavy rainfall (*c*.20 mm) over a 1-2 week period following the winter slurry applications (to grass on 26 Jan and arable on 8 Feb) resulted in peak $NH_4$-N concentrations of 2.8 and 5.2 mg $l^{-1}$ from the arable and grassland plots, respectively. Following the spring slurry application on the 25 April, 10 mm of rain fell 5-6 days after application, resulting in peak $NH_4$-N concentrations of 4.5 mg $l^{-1}$ N from the arable plots, although drainflow volumes were low (0.1 mm). In contrast on the grassland plots, the spring slurry application did not increase $NH_4$-N concentrations in drainage waters, which was most probably a reflection of drier soil conditions (soil moisture deficit = 24mm) under the actively growing grass sward that helped to retain the applied N, compared with 'wetter' soil conditions (soil moisture deficit = 17mm) under the immature winter wheat crop.

Slurry ammonium-N losses up to the time of manufactured fertiliser N application in spring varied between 0.07 and 0.89 kg N $ha^{-1}$ (and accounted for *c*.1% and 8% of measured total N losses from the arable and grassland plots, respectively). In 2003/04, $NH_4$-N losses from the arable (mean 0.08 kg kg N $ha^{-1}$) and grassland (mean 0.10 kg N $ha^{-1}$) plots were similar, with no effect ($P>0.05$) of slurry application timing on $NH_4$-N losses. In 2004/05, $NH_4$-N losses were highest ($P<0.05$) following the winter and spring slurry applications to grassland (0.89 and 0.68 kg N $ha^{-1}$, respectively), compared with losses from the arable plots (all three slurry application timings) and autumn slurry application to grassland (which were in the range of 0.19-0.33 kg N $ha^{-1}$). In 2005/06, $NH_4$-N losses were highest ($P<0.05$) following the winter slurry applications to both the arable and grassland plots at *c*.0.3 kg N $ha^{-1}$, compared with <0.1 kg N $ha^{-1}$ from the other four treatments.

The highest drainflow ammonium-N losses were measured when rainfall (typically > 10 mm) followed within 10-20 days of slurry application onto 'wet' soils (soil moisture deficit < 20 mm). Therefore, in contrast to nitrate leaching losses where the highest risk period for slurry applications was in the autumn, the highest risk periods for ammonium-N losses were in the winter (particularly) and spring.

***Dissolved P losses***

On both the arable and grassland plots, autumn slurry applications had no effect on drainage water total dissolved P (TDP) concentrations in any of the three years (Figure 4). However, as was the case for $NH_4$-N concentrations, there were elevated drainage water TDP concentrations following the winter and spring slurry applications to 'wet' soils where rainfall (typically >10mm) followed, usually within 10 days of slurry application (Figure 4). The highest TDP concentrations were measured following the winter and spring slurry applications, where rainfall events (between 7 and 30 mm) occurred 1-2 days following spreading, although there was no clear relationship between rainfall quantity and peak TDP concentrations.

In 2003/04, there was a small peak in drainage water TDP concentrations (0.25 mg $l^{-1}$) 4 days following the 17 December slurry application on the arable plots and no peak on the grassland plots, reflecting the drier condition of the grassland soil (which had not yet reached field capacity) at the time of slurry application. In March 2004, 20 mm of rain fell 9-12 days after the slurry application and drainage water TDP concentrations peaked at 1.3 and 1.6 mg $l^{-1}$ from the arable and grassland plots, respectively.

In 2004/05, 21 mm of rain 1-2 days following the 16 December slurry application resulted in peak drainage water TDP concentrations of 2.0 mg $l^{-1}$ from the arable plots and 7.3 mg $l^{-1}$ from the grassland plots (Figure 4).

Similarly, 30 mm of rain 6-7 days following the 23 March slurry application resulted in peak drainage water TDP concentrations of 0.68 mg $l^{-1}$ from the arable plots and 5.2 mg $l^{-1}$ from the grassland plots.

In 2005/06, heavy rain (20 mm) 1-2 weeks after the winter slurry applications (to grass on 26 Jan and arable on 8 Feb) resulted in peak drainage water TDP concentrations of 1.2 and 3.1 mg $l^{-1}$ from the arable and grassland plots, respectively. In April 2006, 10 mm of rain fell 5-6 days after the 25 April slurry application, resulting in peak drainage water TDP concentrations of 3.6 mg $l^{-1}$ from the arable plots. However, there was no increase in TDP concentrations from the grassland plots following the spring slurry application (as also observed for $NH_4$-N), which was again most probably a reflection of 'drier' soil conditions (soil moisture deficit = 24mm) under the actively growing grass sward helping to retain the applied slurry P (and $NH_4$-N), compared with the wetter soil conditions (soil moisture deficit = 17mm) under the immature winter wheat crop.

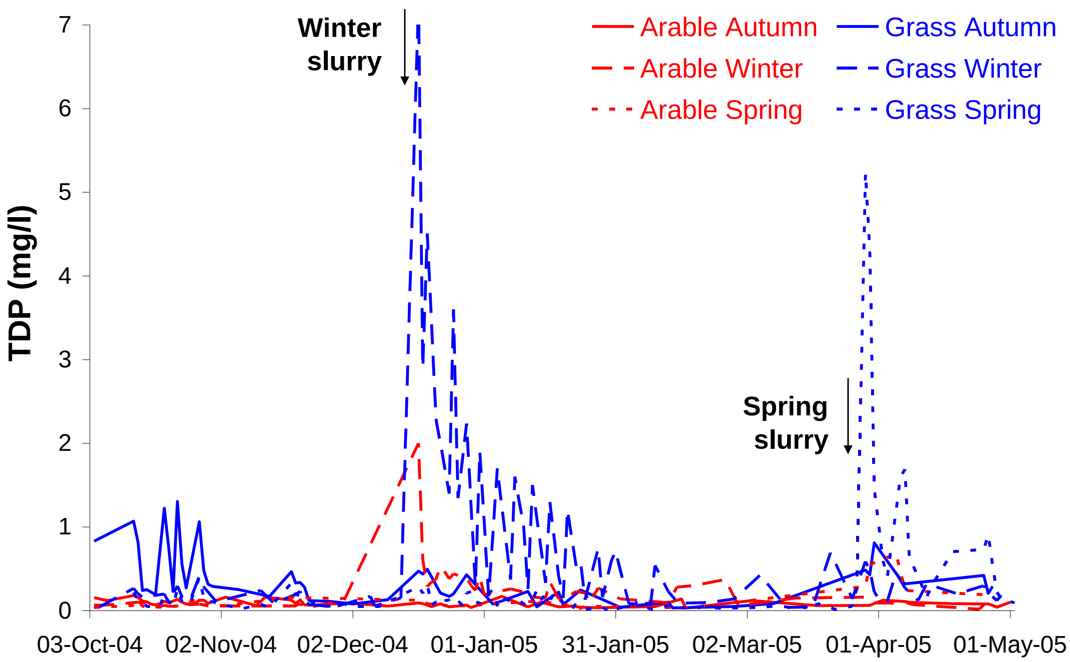


*Figure 4.* Total Dissolved Phosphorus (TDP) concentrations in drainage waters (2004/05)

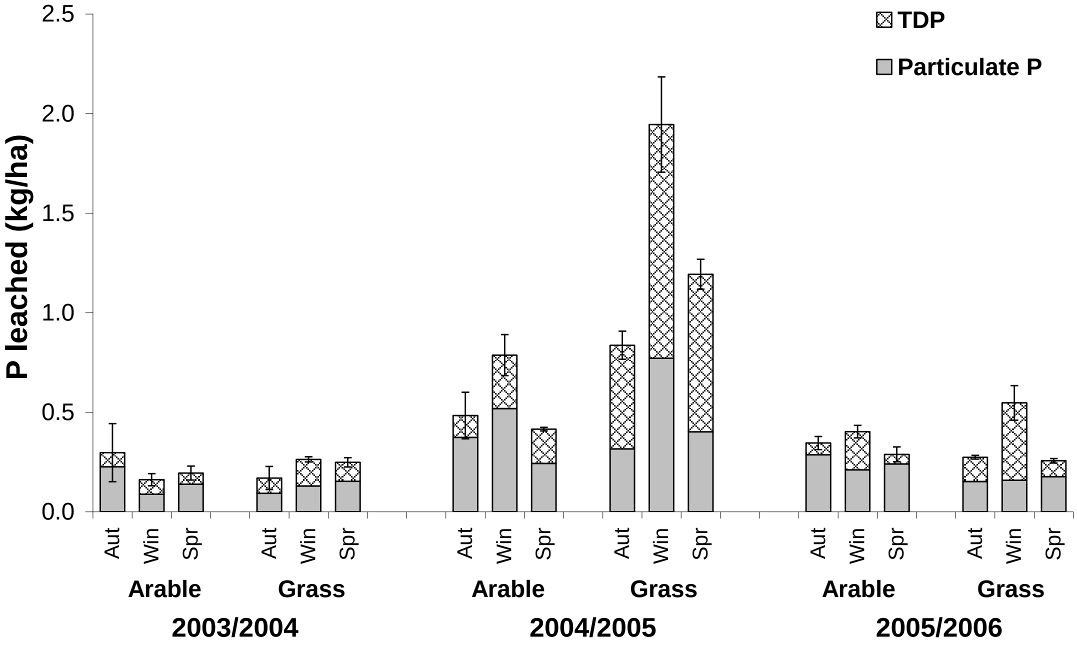


*Figure 5.* Phosphorus losses in drainage waters (2003/04 to 2005/06)

TDP losses (until the end of drainage) from all the treatments varied between 0.05 and 1.17 kg ha$^{-1}$ (Figure 5) and accounted for a mean of 31% and 52% of total P losses from the arable and grassland plots, respectively. TDP losses were greater ($P$<0.05) from the grassland (mean = 0.45 kg ha$^{-1}$; range 0.08-1.17 kg ha$^{-1}$) compared with the arable plots (mean = 0.12 kg ha$^{-1}$; range 0.05-0.27 kg kg ha$^{-1}$) in all three years. TDP losses from the grassland plots were higher ($P$<0.05) following the winter slurry applications than the autumn and spring timings in all 3 years (although the differences were not significant in 2003/04) (Figure 5), reflecting good connectivity between the soil surface and field drains (where the winter slurry applications were made to 'wet' soils) and greater drainage volumes than following the spring slurry timings.

Drainage water TDP concentrations followed a similar pattern to $NH_4$-N concentrations. The highest TDP concentrations and losses (and similarly the MRP data not presented here) were measured where slurry was applied to 'wet' soils and significant rainfall (typically >10 mm) followed, usually within 10-20 days of slurry application (Figure 4).

### ***Particulate and total P losses***

Overall, particulate P losses accounted for a mean of 69% and 48% of total P losses from the arable and grassland plots, respectively (Figure 5). There was no clear effect of land use on particulate P losses in all three years (mean annual losses in range 0.12-0.50 kg P ha$^{-1}$). Similarly, there was no effect of slurry application timing on particulate P losses in 2003/04 or 2005/06. However, in 2004/05 particulate P losses following the winter slurry applications on both the arable land (0.52 kg ha$^{-1}$) and grassland (0.77 kg ha$^{-1}$) plots were higher than those following the autumn/spring slurry applications. The higher particulate P losses following the winter slurry applications were most probably due to particulate matter from the slurry contaminating drainage waters where heavy rain (21 mm) fell 1-2 days after application.

Overall, slurry P losses accounted for 64% and 43% of total P losses following the winter applications on the grassland and arable plots, respectively. Also, overall slurry P losses accounted for 28% and 26% of total P losses following the autumn applications, and 41% and 13% of total P losses following the spring applications on the grassland and arable plots, respectively. Mean slurry total P losses (Figure 6) were highest ($P$<0.05) following the winter applications (compared with the autumn and spring applications) at 0.60 kg total P ha$^{-1}$ (0.45 kg dissolved P ha$^{-1}$) on the grassland plots and 0.21 kg total P ha$^{-1}$ (0.12 kg dissolved P ha$^{-1}$) on the arable plots.

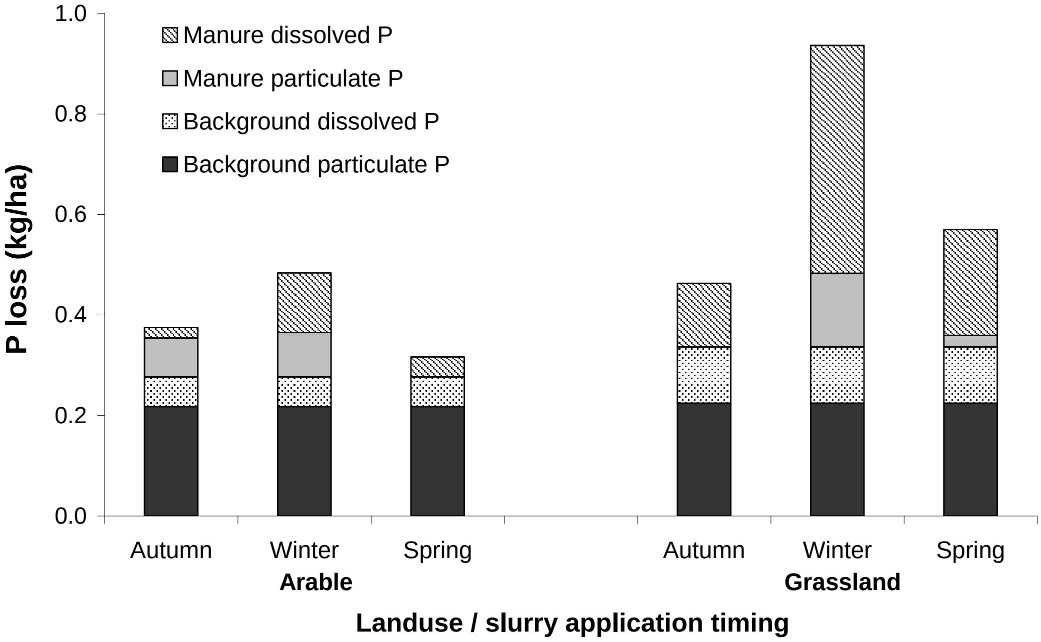


*Figure 6.* Apportionment of phosphorus losses in drainage waters (mean data 2003/04 to 2005/06)

**Discussion**

Previous work has shown that nitrate leaching losses are higher following autumn slurry applications than following winter or spring slurry timing on free draining, sandy/shallow soils (SMITH and CHAMBERS, 1998; CHAMBERS *et al.*, 2000). HAYGARTH *et al*. (1998) and SMITH *et al*. (2001) have also reported that manure applications to drained soils can result in considerable N and P losses to rivers, as a result of by-pass flow to the drains. However, little was known about the processes and conditions that favour the loss of P and ammonium-N to water following slurry applications, and how application timing (in relation to season, antecedent soil moisture and subsequent rainfall) can affect losses.

The results presented in this paper confirm that in terms of slurry application timing and land use, the highest nitrate leaching risk is presented by autumn slurry applications on arable land. Nitrate-N concentrations were greatest following autumn slurry applications to arable land (peak concentrations 20-30 mg l$^{-1}$ greater than from control plots). Notably, the slurry applications had no effect on drainage water nitrate concentrations from the grassland plots; arable reversion to grassland was highly effective at reducing nitrate leaching losses. These data demonstrate the potential effectiveness of arable reversion grassland in reducing nitrate leaching losses from drained agricultural land.

The lower nitrate-N concentrations and leaching losses in drainage waters from the arable reversion grassland plots was probably a reflection of grass N uptake in the autumn and the accumulation of N in the soil organic matter reserves. In autumn 2001 (when the grass was established), the topsoil (0-15 cm) total N content was 0.34%. In autumn 2007 (6 years after the grass was established), the topsoil total N content on the arable reversion grassland plots was 0.41%; 17% higher ($P$<0.001) than on the arable plots at 0.35%. The increased topsoil total N content on the arable reversion grassland plots was probably due to a combination of the lack of cultivation (which would have stimulated N mineralisation) and build-up of organic matter under permanent grassland management, which allowed N to accumulate in soil organic matter reserves. In contrast, annual cultivation of the arable plots would have stimulated the oxidation and breakdown of soil organic matter and mineralisation of N. The reversion of arable land to permanent grassland will significantly reduce nitrate leaching losses by increasing N storage in the soil microbial biomass and organic matter reserves. However, the benefits of reduced nitrate leaching losses must be set against reduced drainage (water resource) volumes of *c.*40% compared with arable cropping.

Drainage water nitrate-N concentrations and total nitrate-N losses (kg $ha^{-1}$) were related to the size of the autumn soil mineral nitrogen pool and autumn/winter drainage volumes. By contrast, slurry ammonium-N and TDP losses were dependent on the antecedent soil moisture conditions and the number of days between slurry application and the next 'significant' rainfall event. The highest drainflow ammonium-N and TDP concentrations were measured where slurry was applied to 'wet' soils (soil moisture deficit < 20 mm) and when typically over 10 mm rainfall followed, usually within 10-20 days of slurry application. Winter and early spring slurry applications therefore pose a greater risk for slurry ammonium-N and TDP losses than autumn timings, because the soils (at or close to field capacity) are unable to retain significant amounts of rainfall and drainflow can occur within a few hours of rainfall commencing.

Phosphorus and ammonium-N are usually strongly adsorbed on to soil colloids and are relatively immobile within soils. Following the autumn slurry applications, there was generally sufficient time for P and ammonium-N to have been adsorbed onto the soil matrix, and for ammonium-N to be transformed within days/weeks to nitrate-N, via the microbially mediated process of nitrification. The elevated TDP and $NH_4$-N concentrations in drainage waters following slurry application to 'wet' soils in winter and spring were a result of these pollutants moving rapidly from the soil surface to field drains, via cracks/mole drains, with little interaction with the soil matrix. There was some evidence (particularly in 2004/05) of higher TDP and $NH_4$-N concentrations and losses from the grassland than the arable plots, which was most probably a reflection of the greater connectivity on the grassland plots between the soil surface and field drains, as a result of 'by-pass' flow in cracks/mole channels, than on the cultivated arable plots.

Overall, slurry P losses accounted for 26-28% of total P losses following the autumn slurry applications, 43-64% of total P losses following the winter timings and 13-41% of total P losses following the spring timings. Also, on the grassland plots, slurry TDP losses from the winter and spring slurry applications were significantly greater ($P$<0.05) than background TDP losses. The spring slurry applications therefore made an important contribution to total P and TDP losses. This has particular relevance for the overall biological impact of P losses, as elevated soluble P concentrations may have a disproportionately large impact in spring and early summer when biological activity in streams and rivers is increasing. Notably, overall TDP losses accounted for a mean of 31% of total P losses from arable plots and 52% from grassland plots. If a large proportion of P losses are present in a soluble form they are likely to have a greater biological impact (HIVLEY *et al*., 2005). Avoiding slurry applications to 'wet' soils in spring (as well as winter) is therefore important from an ecological impact viewpoint.

**Conclusions**

The results from this study on drained clay soils show that spring slurry application timings present the lowest risk and autumn timings the highest risk of nitrate leaching loss. However, winter (and less so spring) slurry applications are likely to pose the greatest risks of elevated $NH_4$-N and TDP concentrations in drainage waters, as during this period soils are typically 'wet' and slurry derived $NH_4$-N and TDP can be transported rapidly from the soil surface to field drains, via cracks/mole channels (an example of 'pollution swapping'). Slurry application timings to soils with a moisture deficit >20 mm (where significant rainfall does not occur in the following 10-20 days) are unlikely to result in elevated drainage water $NH_4$-N and TDP concentrations.

Moving slurry applications from autumn to spring will be effective in reducing nitrate losses to water, but will increase the risk of ammonium-N and P losses to water, unless account is taken of antecedent soil moisture conditions. Farmers will need to ensure that they have sufficient over-winter slurry storage capacity to provide the flexibility to spread slurry when soils are not 'wet' and have dried out sufficiently in spring (i.e. ideally when the soil moisture deficit is >20mm) so that elevated drainage water $NH_4$-N and P concentrations are unlikely to occur. Also, investment in slurry bandspreading/shallow injection equipment will be necessary to facilitate slurry application to growing grassland and arable crops in spring, without compromising crop yields and quality.

**Acknowledgements**

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# SOIL-HYDROLOGICAL MEASURING STRATEGY TO ESTIMATE WATER BALANCES IN THE FLAEMING REGION, GERMANY

*Gundula PAUL*[1] *– Ralph MEISSNER*[2] *– Gregor OLLESCH*[3]

[1] Helmholtz Centre for Environmental Research – UFZ, Department of Soil Physics, Brueckstrasse 3a, 39114 Magdeburg, Germany, gundula.paul@ufz.de

[2] Helmholtz Centre for Environmental Research – UFZ, Department of Soil Physics, Lysimeter Station Falkenberg

[3] Elbe River Catchment Community, Liaison Office, Magdeburg

**Abstract**

Water management often aims at two conflicting goals: On the one hand, humans need to be supplied with high-quality drinking water. On the other hand, a sustainable agricultural production has to be facilitated in the region where groundwater recharge takes place. This paper presents the first results of a project that aims at a better understanding of this problem. Therefore, the Flaeming region in eastern Germany has been selected as a model region. The measurement concept, a bottom-up approach which focuses on the measuring of pedo-hydrological data at different agricultural and forestry used sites and the scaling-up of these data to model realistic scenarios for sustainable land management strategies for the investigated region is described.

These water management activities in the sensitive Flaeming region are embedded in the interdisciplinary joint research project “Sustainable Land Management in the North German Lowland – NaLaMa-nT”, sponsored by the German Ministry of Science and Education (BMBF) from September 2010 until August 2015. The aim of the entire project is to develop a knowledge and decision basis for a sustainable land management concept in four representative model regions in the northern part of Germany (administrative districts Diepholz, Uelzen, Flaeming, and Oder-Spree). The scientific work is divided into 21 subprojects dealing with climatology, water management, agriculture, forestry, and socio-economic aspects.

**Keywords:** Flaeming region, Measurement strategy, NaLaMa-nT, North German Lowland, Redox potential, Soil hydrology, Soil water tension, Sustainable land management, Watermark sensor

## Introduction

Flaeming is a landscape in Eastern Germany (Federal States of Brandenburg and Saxony-Anhalt) that is characterised by its importance for the drinking water supply of approximately 152.000 inhabitants as well as for agricultural production and forestry (BORGMANN, 2012). Due to the limitation of resources, in this region stakeholder conflicts exist. On the one hand, Flaeming, especially Central-Flaeming, is an important groundwater recharge area (LUCKNER, 2002). Consequently, the raw water quantity and quality of the water have to be protected so that a sustainable supply of drinking water to the region according to the principles of the EU Water Framework Directive is warranted. On the other hand, agricultural and forestry production is an economic basis for this rural area and therefore, water is also necessary and essential. The current and future change of climatic conditions, demographical structures, and globalised markets are also influencing the regional water cycle and must be secured for the development of sustainable site adapted agricultural, forestry, and water management strategies.

The before mentioned water management research activities are part of the interdisciplinary joint research project "Sustainable Land Management in the North German Lowland – NaLaMa-nT", which is sponsored by the German Ministry of Science and Education (BMBF) over a period of five years from September 2010 until August 2015. This project aims at the development of a knowledge and decision basis for a sustainable land management concept in a west-east transect in Northern Germany: from the administrative district Diepholz at the Dutch-German border via the administrative district Uelzen and the Flaeming region to the administrative district Oder-Spree at the German-Polish border. The model regions show differences amongst others in natural space (e.g. climate, soils, topography), structure, economical, and demographical structures. However, these regions represent the range of environmental and socio-economic conditions in North Germany.

The scientific work within this research project is divided into 21 subprojects dealing with climatology, water management, agriculture, forestry, and socio-economic aspects. These subprojects are grouped in four cluster (ecological basics, risk management, land use and use of resources, creation of values), which are necessary for the trans- and interdisciplinary cooperation within the research project and efficient transfer of scientific knowledge (Figure 1).

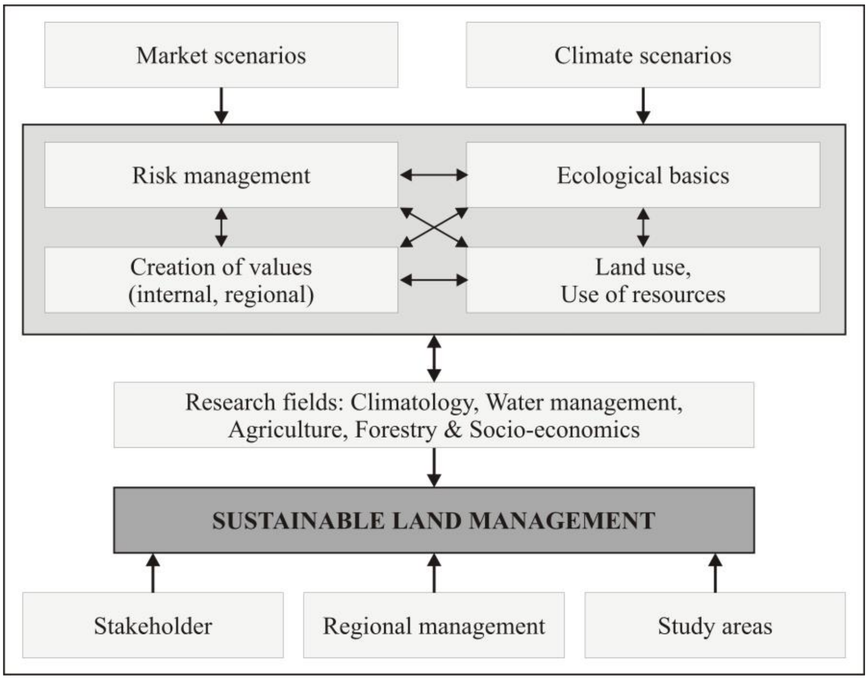


*Figure 1*. Structure of the interdisciplinary research project NaLaMa-nT. SPELLMANN, 2011, modified.

An analysis of the actual conditions forms the basis for further works (Figure 2). Developing scenarios by using climate and market scenarios will be created and adaptation strategies for the model regions will be defined and discussed with stakeholder. Thereby, changing ecological and socio-economic framework conditions like climate change, technical progress, and the globalisation of markets play an important role. Both opportunities and risks might arise for the agriculture and forestry sector during the transition. Structural changes in the model regions require adaptation strategies and – if necessary – realignment of the land management (SPELLMANN, 2011).

In a second step, today's mission statements will be composed and a change analysis as well as an extrapolation of these mission statements will follow (Figure 2). Henceforth, regional options of actions should be extracted for saving the future sustainable development of the rural regions (SPELLMANN, 2011). In cooperation with stakeholder and the regional management authorities of the model regions a sustainable land management concept for the North German Lowland will be developed (Figure 1).

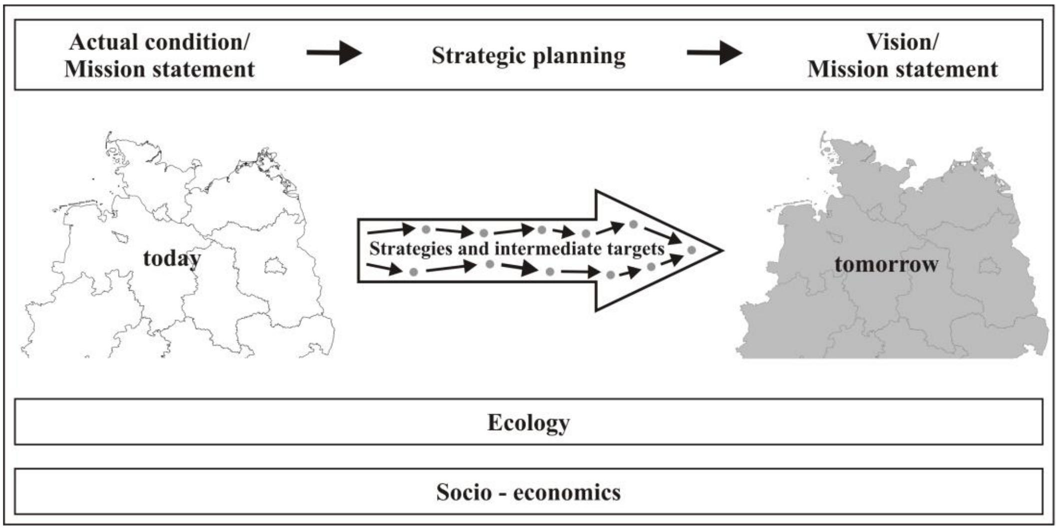


*Figure 2*. Concept of the interdisciplinary research project NaLaMa-nT. SPELLMANN, 2011, modified.
Data basis: © GeoBasis-DE / BKG 2011

This paper focused on the Flaeming region and the objectives are i) the description of the field measurement concept, ii) the presentation of first soil hydrological measurement results and iii) a discussion regarding the scaling-up from point measurement to the entire study area.

**Measurement concept**

An essential aim of the subproject is the estimation of landscape water balances for the representative model regions and their vulnerability during climate and land use changes. In the first stage of this project a field measurement concept for the model region Flaeming was developed. The region is located between Magdeburg in the northwest, Dessau in the south, and Potsdam in the northeast (Figure 3). These measurement results will provide the data basis and will secure process knowledge for model application, which will be done in the second stage of the project. For a successful modelling it is necessary that a plausible representation of the soil moisture in its spatial and temporal patterns exist. Small-scale pedo-hydrological heterogeneities play also a considerable role (SPELLMANN, 2011). For the validation of the model, different land use and soil forms are needed. For this reason seven monitoring places have been established in the model region Flaeming: three on arable lands and four in forest canopies (Table 1, Figure 3). The monitoring places had to satisfy the following criteria:

(1) The most prevalent land use forms in the study area – agriculture and forestry – are investigated. Because the crops rape, maize, and asparagus are typical for the region Rosslau-Wittenberger Vorflaeming, they were selected for this investigation. In forests, for the different forest canopies the compositions and structures of the forests are recorded. In the Central-Flaeming pines are the most important tree species, which cover the largest area in the region (LUCKNER, 2002). Less widespread are deciduous trees (Figure 3), which quota grows in young populations; in this connection amongst others beeches play an important role. Mixed forests occur e.g. in form of beech underplantings under

pines. Thus, different typical land use and soil type (Cambisols, Stagnoslols, Gleysols, Fluvisols, and fens) combinations are investigated.

(2) Because there are known stakeholder conflicts for the water resources, places with a low and high groundwater level have been selected. The different behaviour of the installed sensors can be recorded and possible influences of the water use by drinking water supply, agriculture, and forestry can be recognised. To avoid disturbance of the process, especially in agricultural fields, the entire pedo-hydrological measuring apparatus had to be installed underground.

(3) The pedo-hydrological monitoring places shall be located in the catchment area or close to the catchment of the river Grimmer Nuthe to guarantee a smooth transfer of measured data to model application. The catchment outlet of the river Grimmer Nuthe is defined by the gauge station at the site “Strinum” (Figure 3), for which a long term time series regarding runoff and relevant water quality parameters exist.

*Table 1*. Instrumental equipment at the pedo-hydrological monitoring places in the Flaeming region. (x = installed at this place, – = not installed at this place)

| Land use form | Landscape (according to Brunner 1962) | Watermar k sensors | Redox sensors | Suction plates |
|---|---|---|---|---|
| Asparagus | Rosslau-Wittenberger Vorflaeming | x | – | – |
| Maize | Rosslau-Wittenberger Vorflaeming | x | – | – |
| Rape | Rosslau-Wittenberger Vorflaeming | x | x | x |
| Pine monoculture (low groundwater table ) | Central-Flaeming | x | – | – |
| Beech underplanting under Pine (low groundwater table) | Central-Flaeming | x | – | – |
| Mixed oak forest (high groundwater table) | Central-Flaeming | x | x | x |
| Pine (high groundwater table) | Central-Flaeming | x | - | - |

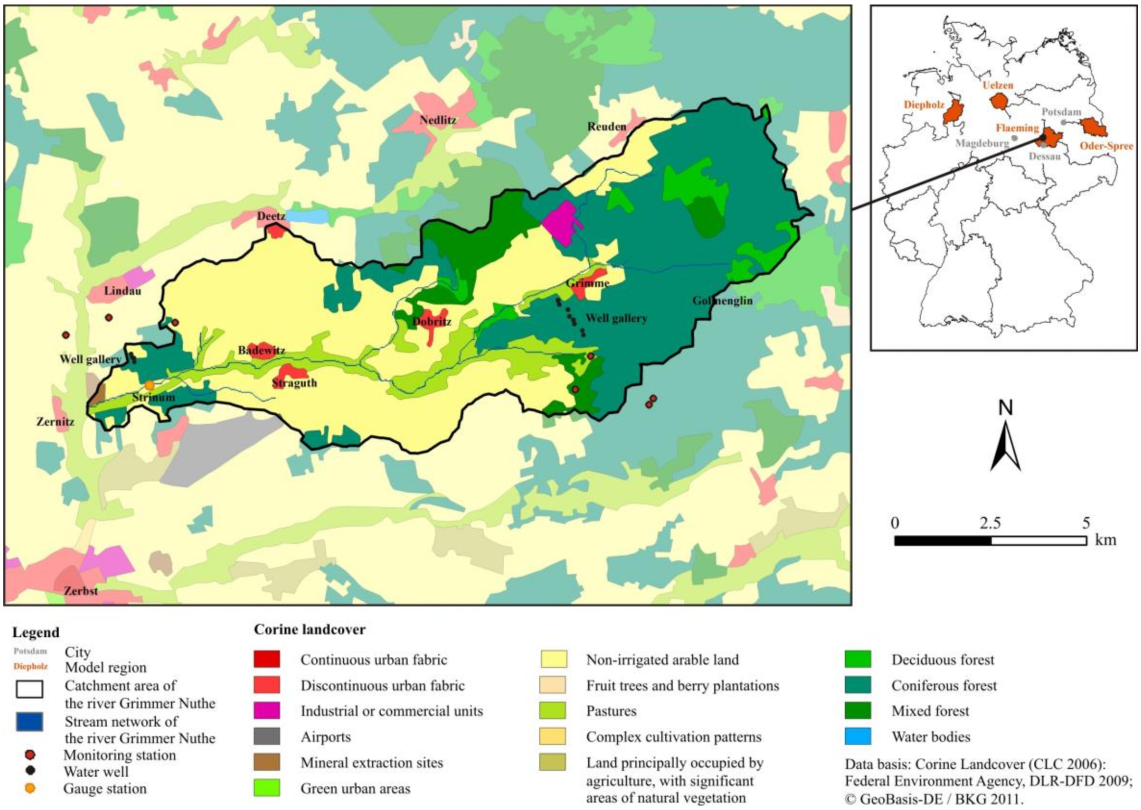


*Figure 3*. Geographical position of the investigated administrative districts in the North German Lowland (right) and detailed information regarding the position of the pedo-hydrological monitoring places in the catchment area of the river Grimmer Nuthe (Flaeming region).

At all locations Watermark Soil Moisture Sensors Model 200SS (Irrometer Company; Figure 4; briefly called Watermark sensors) have been installed at different depths. These sensors are – compared to other resistor principle sensors – relatively favourable and simple to install. Another advantage is that the soil at the measurement site is not destroyed during the installation (SHOCK *et al*., 1998). Furthermore, they withstand low temperatures and require a minimal maintenance (PERTOLL, 2008). With the Watermark sensors the soil moisture tension in cbar is measured; the measuring principle is based on the electrical resistance between two electrodes (LARSON, 1985; HAWKINS, 1993). The operating mode of Watermark sensors is similar to that of gypsum blocks (SHOCK *et al*., 1998; PERTOLL, 2008). The measurement range of Watermark sensors is from 0 to 239 cbar (Irrometer Company 2010), which is equivalent from pF 0.00 to 3.38. The watermark sensors allow the detection of changes and differences in the soil moisture in a high temporal resolution (15-minutes-values). Different and typical drying-out as well as re-wetting processes and percolation into deeper soil layers for the respective land use form can be investigated.

The measurements with the Watermark sensors are supplemented by measurements with a mobile TDR sensor (Fieldscout TDR 100, Spectrum Technologies Ltd.; Figure 4) on those days when the data loggers are read. With the TDR sensor the absolute water content in an integral of 0.2 m under the surface of terrain can be measured. So it is possible to record the spatial variability of the soil water content in the field.

At two places – in a rape field near the town Lindau and in the mixed oak forest near the village Grimme – at this time twelve redox sensors were installed at different depths (Figure 3 & 4). Thereby, the redox sensors and a reference electrode were permanently installed in the undisturbed soil. The voltage difference between the redox sensors (platinum electrodes) and the reference electrode (KCl) can be measured. The advantage of platinum is that it reacts very fast with reaction of electrons and it is in comparison with the other materials mechanically highly loadable (FIEDLER & FISCHER, 1994). In order that the reference electrode does not dry out, a constant flow of the electrolyte solution is ensured.

By measuring the redox potential in soils it is possible to characterise parts of the water balance (MANSFELDT, 1993). MANSFELDT, (1993; 2003) has shown that there are relationships between the measured redox potential and the fluctuating groundwater table in soils. Furthermore, he found that in a Calcaric Gleysol the redox potential has low values if water saturation in the soil exists. If the duration of the water saturation persists longer, the redox potential values are negative. As soon as the groundwater level falls below the installation depth of the redox sensor, the redox potential values increase rapidly. Against this background we want to detect the fluctuations of the capillary fringe by measuring the redox potential at the two monitoring places. In 15-minutes intervals the redox potential values are logged. The redox potential reacts rapidly with changes in soils so that long-term measurements under field conditions in a high temporal resolution are necessary for making significant conclusions (FIEDLER & FISCHER, 1994; MANSFELDT, 2003).

At the same two places suction plates of 0.12 m in diameter have been installed at different depths. By creating a vacuum with a pump it is possible to extract soil water on an event basis in situations of high soil moisture and to analyse these samples in the laboratory. Hence, statements about the dislocation of nutrients in soils will be possible.

Furthermore, two rain gauges (Pluvio Standard Ott, Kempten) have been installed in the study area: first near the town Lindau (Rosslau-Wittenberger Vorflaeming) and the second near the village Golmenglin (Central-Flaeming; Figure 4). With the measured precipitation amounts (five-minutes-values) it will be possible to analyse the interactions between precipitation and the soil moisture.


*Figure 4*. Essential measuring instruments used at the pedo-hydrological monitoring places in Flaeming; top left: Watermark sensor with data logger; top right: redox sensors; bottom left: mobile TDR-sensor; bottom right: rain gauge.

**Model approach and scaling-up**

Based on the gathered process knowledge the measurement results will be used for an adaptation and further development of the model IWAN (**I**ntegrated **W**inter **E**rosion **A**nd **N**utrient **L**oad **M**odel; OLLESCH *et al*., 2006). This model enables the integration of the different pedo-hydrological measuring plots and allows after the validation of the site conditions the calculation of scenarios for best management practices in the region. The validation of model results will be conducted for the 85.7 km² catchment Grimmer Nuthe, in which the measurement plots are located. This also allows a proper definition of model parameters and variables.

Similarly, these results from physically based hydrological modelling are an essential input in the validation procedure for calculating a landscape water balance on the regional scale for the four model regions of the project, which vary in size from 1454 to 2243 km² (SPELLMANN, 2011). For these estimations the method of Bagrov and Glugla (BAGLUVA-Method) and the TUB-BGR-Method (Figure 5) will be used. Both methods are established procedures in Germany and recommended for practical application, e.g. in the Hydrological Atlas of Germany (BMU 2000). The methods are described in detail by GLUGLA, *et al*. (2003) and WESSOLEK, *et al*. (2008). The results of both methods as well as of the west-east transect of the four model regions will be compared and evaluated. The landscape water balances will be calibrated and verified for the period from 1990 until 2010.

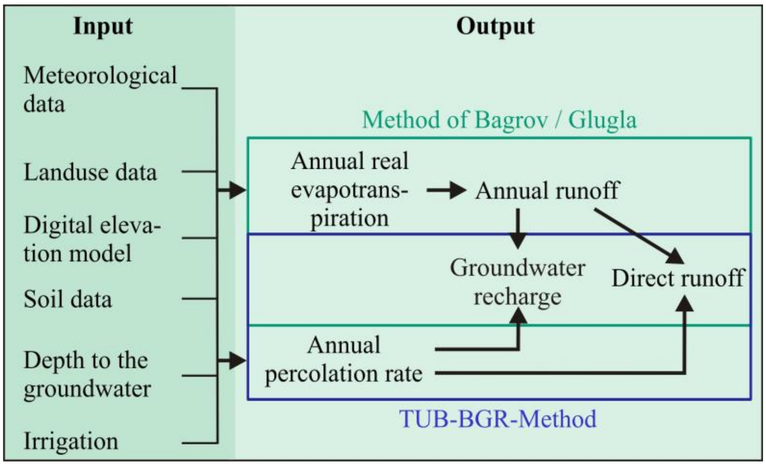


*Figure 5*. Used model approaches for the calculation of landscape water balances.

Future development and vulnerability of single water balance elements will be estimated with different realisations from the new IPCC-scenarios RCP 8.5 (**R**epresentative **C**oncentration **P**athways, powered by the atmospheric general circulation model ECAHM 6), which reflect probable extremes in climatic water balances.

**First results and discussion**

First results of the monitoring that started in spring 2011 showed that small-scale dynamics in the soil moisture at one place as well as at local differences are visible and the influence of land use can be recorded. In Figure 6 two examples are shown. At both places, approximately 100 m apart, there are no significant differences in the soil characteristics but in the soil water dynamics, which is triggered by the different vegetation. The recorded soil moisture tensions in the beech underplanting under pine (Figure 6b) are higher than in the pine monoculture (Figure 5a), especially after the leaf bloom of the beeches at the beginning of May. As soon as the measured soil moisture tension values are higher than 30 cbar (= pF 2.48), which conforms with the lower limit of the field capacity, at both places a day-night-rhythm was visible, which is caused by evapotranspiration. This indicates a drought stress in the upper soil layers for the vegetation. In the main root zone of the beech underplanting under pine (sensor depth 0.5 m; Figure 6b) the soil moisture tension values are higher than in the pine monoculture.

Until June 20 the drying-out processes continue due to extreme dry weather situation in spring 2011 (DWD 2011). The soil moisture values approached the permanent wilting point. After some smaller precipitation events around this time, the soil became wetter and the soil moisture tension values decreased. As a consequence of several precipitation events in the summer and autumn the soil did not dry out to the same extent as in spring. After each precipitation event a re-wetting of the soil took place and the soil moisture tension values decreased (Figure 6a & b). During the precipitation period the soil at a depth of 0.5 m was moisturised earlier than in a depth of 0.1 m. The sensors in the upper soil layers are installed within the litter layer, which is water repellent. Consequently, the soil moisture tensions did not changed until the soil (respectively the litter layer) was completely saturated. On the contrary, the sensors in the deeper soil layers are installed in sandy soil so that the soil became wet and a decrease in the soil moisture tensions occurred earlier.

The soil moisture tension values in the beech underplanting under pine have been higher than in the pine monoculture during the entire period until November (Figure 6a & b) which was caused by larger canopy transpiration in pine with beech underplanting under pine and above average air temperatures in November 2011 (DWD 2011). However, we expect that the differences regarding the soil moisture tension values between the beech underplanting under pine and the pine monoculture will decrease in the course of December because the active evapotranspiration period of deciduous forests ends in November (ZIMMERMANN *et al*., 2008).

Unlike the upper soil layers for the pine monoculture to a depth of 1.05 m, no great differences regarding the soil moisture tension values over the measurement period have been registered (Figure 6a). Only in the very dry spring period 2011 did the soil dry out marginally at this depth, but the soil moisture tension values were still in the range of saturation. In the first half of July, an amount of about 145 mm precipitation was measured so that the entire soil column became saturated. Although, there were several precipitation events from July until the end of October, no direct influence of the precipitation events on the soil moisture tension in this depth has been observed. A small increase of the soil water tension values has been registered.

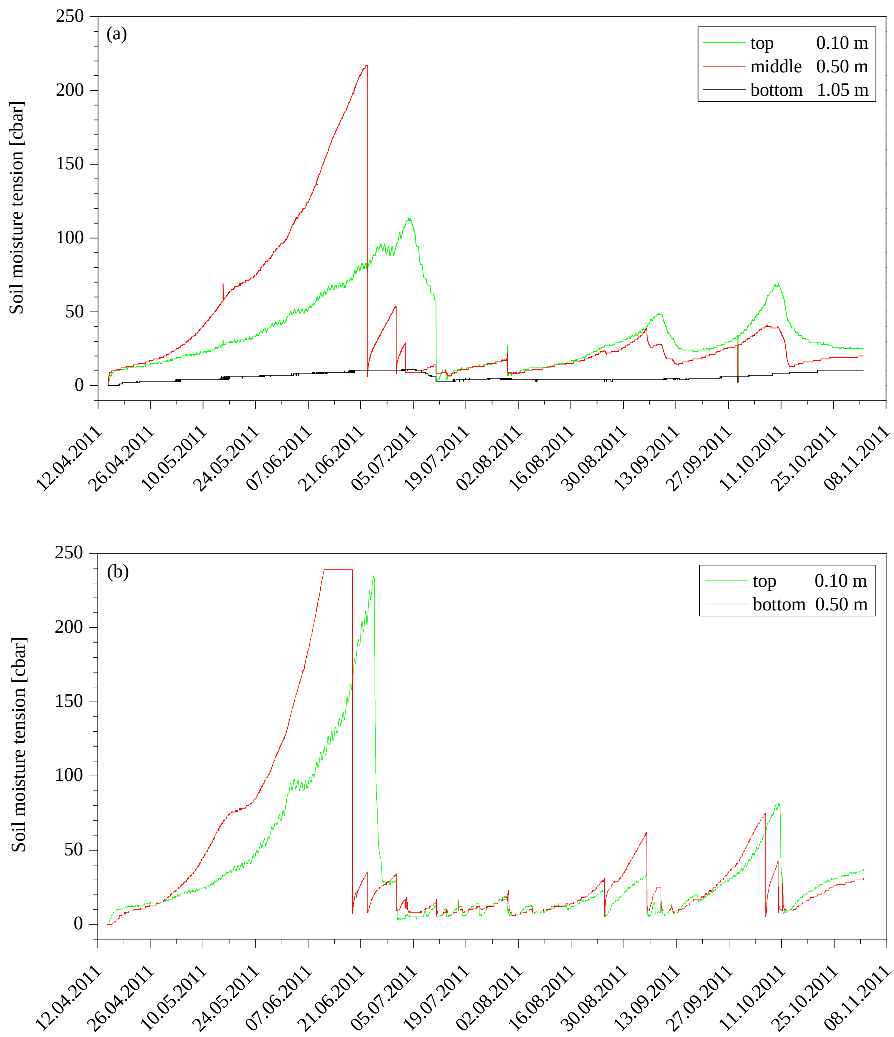


*Figure 6*. Examples of the development of the soil moisture tensions at two closly neighbouring places with a low groundwater level in a forest; a: pine monoculture, b: beech underplanting under pine.

First measurement results concerning the redox potential are shown for data of a mixed oak forest stand, which is characterised by a high groundwater level (Figure 7). The installation of the redox sensors took place at September 22, 2011. The influence of the soil disturbance during sensor installation is only visible on the first days of recording until September 26, 2011. The outliers at September 29 and October 27, 2011 are caused by some irregularities during the change of the accumulator. During the whole measuring period the redox potential values at 0.20 and 0.60 m have been positive and at 0.85 and 1.40 m have been negative. In accordance to MANSFELD (1993) these results indicate that the upper limit of the capillary fringe must be moved between 0.60 and 0.85 m soil depth. Because there was no precipitation measured at this site in the period until October 6 2011, the redox potential values increased.

8, 12, and 18 mm precipitation have been measured on October 6, 10 and 11 2011, respectively. As a result, the redox potential values decreased as the soil became wet again. In the last two weeks of October 2011 daily precipitation occurred, but the daily amounts were usually less than 1 mm. Nevertheless, a slight decrease of the redox potential has been observed, especially in the deeper parts of the soil. It should be mentioned that the upper parts of the soil column are additionally influenced by atmospheric conditions, i.e. higher temperature which accelerate possible biogeochemical reactions. At the beginning of October 2011 the air temperature decreased and, consequently, the redox potential signals in the upper sensors approach a constant level. It is remarkable, that during the whole measuring period, the fluctuating water table influenced the upper as well as the lower soil layers.

At the aforementioned mixed oak stand there is also a distinctive daily pattern of the redox potential values (Figure 8). In the morning hours the redox potential decreased and the minimum is reached between 8.30 and 9.00 a.m., approximately one hour after sunrise. During the day the redox potential increased and the maximum is reached between 5.30 and 6.30 p.m., approximately one hour before sunset. Therefore, the daily maxima and minima occur in the depth with a short temporal delay. The observed pattern of the redox potential is similar to that of daily variations in soil temperature, which are known. If the soil temperature increases, less oxygen is solute in the water and, thus, the redox potential decreases as DUŠEK et al. (2008) described. However, the persistence of these daily fluctuations also in deeper measurement levels gives proof of other external influences in the mixed oak forest stand, which is characterised here. Obviously, the vegetative rhythm of the oaks and the ground vegetation layer underneath has an influence on the redox potential fluctuations of the soil on this short daily term. Triggered by transpiration, the roots change the soil pore water milieu either by root exudates or changes in soil water solute concentrations by increasing the soil water tension. Similar day-night-rhythms that support these hypotheses are typical for the soil moisture tension measurements described above. These details in the measurements, which are conducted so far, show that the measuring apparatus can detect properly even small and short term variabilities.

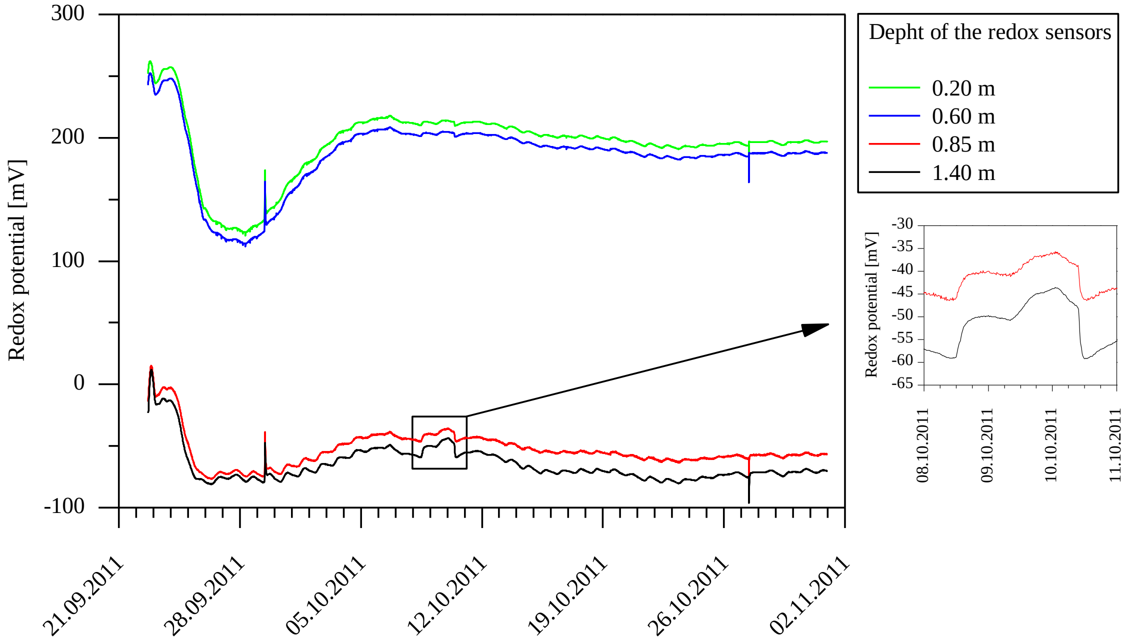


*Figure 7*. Development of the redox potential in a mixed oak forest stand (low groundwater table).

However, there is one situation in the redox measurements at the lower soil horizons that does not fit exactly into the above characterised day-night temperature regularities. On October 9 and 10, 2011 the sensors at 0.85 and 1.40 m depth show slightly higher values but do not overprint the day-night-rhythm (Figure 7, small diagram). This may be caused by lateral water flow and mixing of local pore water with lateral water of different chemistry after rainfall on these days. This again gives evidence that the expected prompt reaction of the installed measurement system is highly probable and it is able to give detailed site specific pedo-hydrological information from the investigated region.

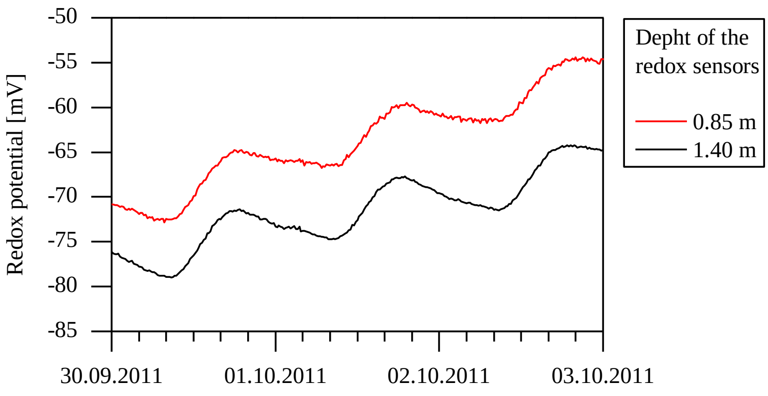


*Figure 8*. Daily fluctuations of the redox potential in 0.85 and 1.40 m depth in a mixed oak forest stand (low to groundwater table).

## Conclusion

The structure, partners and the applied methods in the joint research project NaLaMa-nT are presented in this paper. The project aims at knowledge development and transfer from science to practical application. The measurement system that is installed in one of the four model regions of the project allows the identification of processes which are essential for the water balance, the short term variability and the small-scale heterogeneity, which are driving forces for the water flow in this typical landscape of Northern Germany. The results form a basis for model application and scenario calculation regarding the impact of climate change impacts on regional water balance. It is expected that the outcomes are important for the future management of the water resources in the regions.


## Acknowledgements

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# WATER QUALITY CHANGES FOLLOWING INTENSIVE FOCUS ON MITIGATION METHODS TO REDUCE PHOSPHORUS LOSSES IN THE CATCHMENT OF LAKE VANSJØ, NORWAY

*BECHMANN, M.*[1] *– ØGAARD, A.F.*[1]

[1] Bioforsk, Norwegian Institute for Agricultural and Environmental Research, Fred.A.Dahlsvei 20, 1432 Aas, Norway, E-mail: marianne.bechmann@bioforsk.no

**Abstract**

The objective of this paper was to evaluate the effect of the comprehensive integrated effort that has been put into reducing diffuse P losses from agricultural areas around the lake western Vansjø, a pilot area within the EU's WFD.

The mitigation methods consisted of reduced tillage, reduced P fertilizer application, grassed buffer strips and constructed wetlands. In addition, sewage treatment systems were established for private households to reduce P outlet. The strategy to implement mitigation methods consisted of, farmer meetings with discussions, field trips, environmental planning on individual farms and last but not least contracts with economic incentives.

Monitoring of the water quality in six small streams consisted of manual water sampling carried out bi-weekly and during runoff events from May 2005 and to April 2011. The streams represented runoff from three catchments with intensive potato and vegetable production and three catchments dominated by cereal production.

Results showed that from 2004 to 2010 the use of P fertilizer had been reduced by approx. 75 %. The area of no-till in autumn increased and for some of the catchments covered 100 % of the area in 2010. Vegetated buffer strips were established along streams in three of the catchments and 3 constructed wetlands were built during the period 2005-2011.

The concentrations of TP and SS were significantly influenced by the amount of runoff and no significant effects of mitigation methods on TP and SS losses were detected. However, there was a tendency of a decrease in the TP/ SS relationship during the monitoring period. The results indicate that the large reduction in P fertilizer application together with improved sewage systems explained part of the decrease in concentration of TP relative to the SS concentration for the six agricultural streams in this study.

The operational monitoring was not able to show significant changes in TP concentrations, despite the large investments in mitigation. Targeted monitoring over a long time is required in order to detect effects of mitigation methods implemented at the catchment scale.

**Keywords:** Agriculture, erosion, fertilizer, vegetated buffer, constructed wetlands, sewage, suspended sediments

**Introduction**

Water quality problems in the EU countries as in other countries around the world are substantial in areas with intensive human activities (EEA, 2005). Contributions from wastewater and agriculture are often dominating sources of nutrients causing eutrophication (JARVIE *et al.*, 2010; ARNSCHEIDT *et al.*, 2007). Both nitrogen and phosphorus (P) contribute to eutrophication, but in most freshwaters P has been shown to be the limiting factor (FOY, 2005). Mitigation methods to reduce P loading to surface waters have been implemented during the last two decades in European countries, especially after implementation of the EU's water framework directive (SCHOUMANS *et al.*, 2011).

Despite the focus on removing point sources (SHARPLEY and TUNNEY, 2000; WITHERS *et al.*, 2000), losses of nutrients from sewage systems from private households in the agricultural landscape are not totally eliminated and still contribute to impaired water quality, especially during low flow periods (Withers *et al.*, in press). Hence, both point and non-point source of P should be targeted.

Diffuse losses of Pare, compared to point sources, are difficult to mitigate. In artificially drained soils, the source areas are widespread in the landscape (BECHMANN and DEELSTRA, 2006). Additionally, the effects of mitigation methods are influenced by both weather and agricultural land use changes, e.g. intensification of agricultural production (JARVIE *et al.*, 2010), or they may be masked by other P loss processes occurring in the landscape. In order to increase the chance of success in mitigating diffuse P losses, a holistic set of mitigation methods that involves stakeholders should be implemented (SHARPLEY *et al.*, 2009).

At the same time, strategies to mitigate P losses should evaluate the bioavailability of P losses, since P derived from different sources may give a different response in the lake (FOY, 2005). Erosion and loss of particulate P (PP) are an important process in arable production systems (ULÉN *et al.*, 2010). Changing tillage practice from autumn ploughing to reduced tillage, direct drilling or spring tillage is the most widely used mitigation method to target soil erosion and PP losses at the field scale, but in some cases loss of dissolved reactive P (DRP) may increase by introducing these tillage methods (SHARPLEY and SMITH, 1994). Therefore, it has been argued that soil erosion even contributes to improved water quality by reduced algal growth (EKHOLM and LEHTORANTA, 2012) and tillage methods to reduce PP transfer may be less important for water quality than mitigation methods such as reduced P application.

In areas with horticulture or high intensity livestock production, high soil P status causes high risk of P losses (e.g. JARVIE *et al.*, 2010; KLEINMAN *et al.*, 2011; BECHMANN and ØGAARD, 2010). Losses of P from high P soils have a high bioavailability (Maguire *et al.*, 2005). Other sources of P with high bioavailability in the agricultural landscape are related to effluents from sewage systems or livestock production. Mitigation methods to reduce the most bioavailable P fractions include reduced application of P in fertilizer and manure to reduce soil P content and improving sewage systems (KLEINMAN *et al.*, 2011; NEAL *et al.*, 2010).

The effect of mitigation methods to reduce P losses is well-documented at the plot scale (e.g. soil tillage), hill slope scale (e.g. grassed buffer strips) and in the stream (e.g. constructed wetlands) (SIMS and KLEINMAN, 2005; SCHOUMANS *et al.*, 2011). However, only a few studies have been able to demonstrate an effect of mitigation methods at the catchment scale (SHARPLEY *et al.*, 2009).

The objective of this paper is to evaluate the effect on water quality of a large decrease in P application on agricultural fields, reduced soil tillage in autumn and removal of contributions from point sources around the lake western Vansjø, based on water quality monitoring of 6 small streams contributing to the lake.

**Background**

The lake Western Vansjø in south eastern Norway is eutrophic due to large phosphorus loads to the lake. An objective is to reduce total P concentrations to 50 µg/l in rivers and streams contributing to the lake. Both point and non-point sources of P were identified in 2000 at the outset of the project (LYCHE SOLHEIM *et al.*, 2005). Efforts have been made to improve the water quality since 2000, but during the first years the implemented mitigations were insufficient to improve the water quality. Therefore, in collaboration with different stakeholders, an action plan for reduced P loads was made in 2007. The action plan proposed to set limits for the farming practices in the catchment. By a project funded by the Ministry of Food and Agriculture in Norway, local authorities, agricultural advisors, farmers and research scientists collaborated to reduce the P losses from agricultural areas. The 40 farmers in the catchment were involved in fulfilling the action plan through participation at dialogue-meetings and environmental planning on individual farms. .

The farmers were encouraged to sign an environmental contract including subsidies for covering extra costs and possible less income. The contracts include:

- Use of less P fertilizer than the national recommended level
- No soil cultivation during autumn
- No cultivation of potatoes or vegetables on fields frequently flooded
- Establishment of eight-meter wide grassed buffer strips along open water
- Establishment of grassed waterways where there is a large risk of erosion
- Establishment of constructed wetlands where this is recommended

An agricultural advisor was responsible for the dialogue with farmers and 29 out of the 40 farmers signed the environmental contracts during spring 2008. Therefore a comprehensive implementation of mitigation methods from 2008 was expected.

**Materials and methods**

***Case study area***

The western Vansjø catchment area is 68 km$^2$, including the 12 km$^2$ lake surface area. The agricultural area constitutes 11 km$^2$ and the rest is forest, roads and housing areas. The precipitation normal is 829 mm at Rygge meteorological station (DNMI) and the soil consists of marine clay deposits and moraine deposits. From 2004 to 2011, monitoring of water quality was carried out in 6 agricultural streams contributing nutrients to the western part of Lake Vansjø. (Figure 1). The catchments of these streams are from 13 to 478 ha in size and the agricultural area constitutes from 12 to 90 % of the catchment areas (Table 1). Three of the catchments are dominated by clay loam soils with cereals production, whereas the three other catchments are dominated by sandy loam soils with production of potatoes and vegetables in addition to cereals. Most agricultural areas are artificially tile drained. Animal production is limited, though pig production in Augerød and chicken production in Guthus obviously influenced the soil P status in agricultural soils of these two catchments.

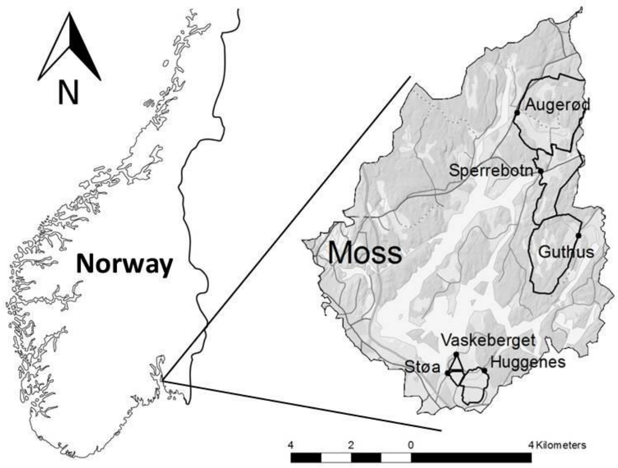


*Figure 1.* Location of the study area in Norway

*Table 1.* Total catchment area, % agriculture and production. Average P-AL values in each of the catchments, minimum and maximum values in parenthesis (n = 3-7 samples/ha)

| Catchment | Total area | Agriculture | Production | Soils | Erosion risk** | Soil P status |
|---|---|---|---|---|---|---|
| | ha | % | | | kg/ha/yr | P-AL*** (mg/100g) |
| Augerød | 478 | 20 | Cereals pigs* | Clay loam and silt loam | 680 | 11 (3-25) |
| Guthus | 315 | 12 | Cereals, chicken | Clay, silty loam and org. | 420 | 21 (2-39) |
| Sperrebotn | 248 | 19 | Cereals | Silt loam and clay loam | 670 | 8 (5-23) |
| Støa | 16 | 89 | Cereals, vegetables, potato, gras | Sandy loam, and silt loam | 420 | 18 (15-20) |
| Vaskeberget | 13 | 91 | Cereals, vegetables, potato | Sandy, silty and clay loam | 480 | 16 (11-20) |
| Huggenes | 81 | 85 | Cereals, vegetables, potato | Sandy, silty and clay loam | 530 | 20 (9-38) |

*no pigs after 2006 **at autumn ploughing *** P-AL values of 0-4 are classified as low, 5-7 as intermediate, 8-14 as high and >14 as very high

Soil P status in the catchments is measured as ammonium acetate lactate extractable P (P-AL; EGNÉR *et al.*, 1960). The average P-AL value is characterized as very high in the three catchments (Støa, Vaskeberget and Huggenes) with production of potatoes and vegetables, because of a high P application to potatoes and vegetables (Table 1). The catchment (Guthus) with chicken production and thereby application of P rich manure, also has a very high average soil P status. The area is relatively flat (1-3 %) and generally has a low erosion risk. However the areas with potato and vegetable production may be even more susceptible to erosion after harvest than ploughed soils, because of soil compaction and damage to soil structure during harvest, and thereby reduced water infiltration capacity.

***Phosphorus application and sewage systems***

Data on P fertilizer application was collected from the farmers with environmental contracts for the years 2004 and 2007-2010. Only a part of the agricultural area had contracts. The part of the area included was: Guthus 100 %, Sperrebotn 92 %, Augerød 75 %, Støa 70 %, Vaskeberget 0 % and in Huggenes 46 %. Consequently, data on P application to agricultural areas in the catchments are only complete for Guthus. Farmers in the catchments with potatoes and vegetable production had lower participation in contracts than the farmers with cereal production only.

Information about sewage treatment systems for private households came from the municipalities (Våler and Rygge).

***Water quality sampling***

The case study area is part of a Pilot area within the WFD and the in-stream monitoring of water quality is an operational monitoring carried out to be able to assess any changes in the status resulting from the programmes of measures.

Monitoring of the six streams consisted of manual water sampling carried out bi-weekly and during high flow events starting 18. October 2004.

Data on concentrations are presented for the years from 1 May to 1 May in order to evaluate the effect of one growing season on the water quality of the following year. The event-based sampling was strategic with the aim of catching the top of flow and representing as many high flow events as possible within the project limits. The number of samples each year is shown in Table 2. There is a risk of missing concentration peaks between the samplings, and therefore number of samples is important for the statistical significance of the results. MOOSMANN *et al.* (2005) estimated 30 as the number of samples required to document significant effects. The number of samples required depends on the variation in concentrations. TP and SS were expected to vary more according to water flow than DRP, and were therefore analysed more frequently.

Samples were transported directly to the laboratory for cooling and analyses were carried out on the same or the following day. Results from one sample in Sperrebotn in September 2010 were left out of the analysis because it had a very high TP concentration (410 µg $L^{-1}$) and not a correspondingly high SS concentration (15 mg $L^{-1}$). Unfortunately this was not caught by the quality control in time to reanalyze the sample.

*Table 2.* No of samples for total phosphorus (TP), dissolved reactive P (DRP) and suspended sediment (SS) each year

| | No of samples | | | | | | |
|---|---|---|---|---|---|---|---|
| | 2004/05 | 2005/06 | 2006/07 | 2007/08 | 2008/09 | 2009/10 | 2010/11 |
| No of samples (TP) | 12-14 | 15-17 | 38-46 | 36-41 | 23-31 | 19-25 | 14-26 |
| No of samples (DRP) | 12-14 | 11-14 | 9-13 | 2 | 8-12 | 7-10 | 4-9 |
| No of samples (SS) | 0 | 0 | 38-46 | 36-41 | 23-31 | 19-25 | 14-26 |
| Runoff (mm) | 331 | 448 | 745 | 660 | 601 | 486 | 491 |
| Runoff station | Skuterud | | Guthus | | | | |

***Hydrology***

For the years 2004/05 and 2005/06, water discharge data from the Skuterud stream (about 25 km from W. Vansjø) is used for water discharge in the streams (DEELSTRA *et al.*, 2005). From the summer 2006 to 2011water discharge was based on water table measurements in a natural transect in Guthus stream. Average runoff was 537 mm/yr for the period 2004-2011.

***Chemical analyses***

All water samples were analyzed for concentration of total phosphorus (TP) in the first year and for selected samples the other years. Samples were also analyzed for concentration of dissolved reactive phosphorus (DRP). In addition, from 2006 onwards, samples were analyzed for concentration of suspended sediments (SS).

Unfiltered samples were used to determine TP by digestion with $K_2S_2O_8$ and filtered samples (< 45 µm) were analyzed for dissolved reactive P (DRP). Phosphorus in all filtrates and neutralized digests was analyzed spectrophotometrically by the ammonium molybdate blue method of MURPHY and RILEY (1962). Suspended sediments were determined by filtering an exact sample volume of 25 to 250 ml after thorough mixing (containing at least 5 mg SS) through a pre-weighed fiberglass filter (Whatman GF/A).

**Results and discussion**

***Implementation of mitigation methods***

In the catchments Augerød, Guthus and Sperrebotn, soil tillage in autumn was considerably reduced from 2004 to 2010, and was omitted during the last two years (Figure 2). Tillage practices in Støa, Vaskeberget and Huggenes also partly changed to no till in autumn on areas with cereals, and constituted in 2010 71 %, 31 % and 53 % of the area in Støa, Vaskeberget and Huggenes, respectively.

Phosphorus application has been considerably reduced from 2004 to 2010 (Table 3). Unfortunately, farmers with intensive vegetable production (Støa, Vaskeberget and Huggenes) were less willing to participate in the mitigation project than farmers producing cereals (Augerød, Guthus and Sperrebotn). The economic benefits of potato and vegetable production are much higher and the economic risk of reduced yields makes it less attractive to reduce P fertilizer application for these crops. However, the establishment of demonstration sites with lower P application rate than common practice in this area were likely to have influenced these farmers as well. Phosphorus application in 2004 reflects the common fertilization practice before focus on reduced P application. For the contracted area, the total P application was reduced by 12 tons from 2004 to 2010.

Vegetated buffer strips (8-12 m wide) were established in Augerød, Guthus and Sperrebotn along 60 %, 100 % and 65 % of the streams in these catchments, respectively.

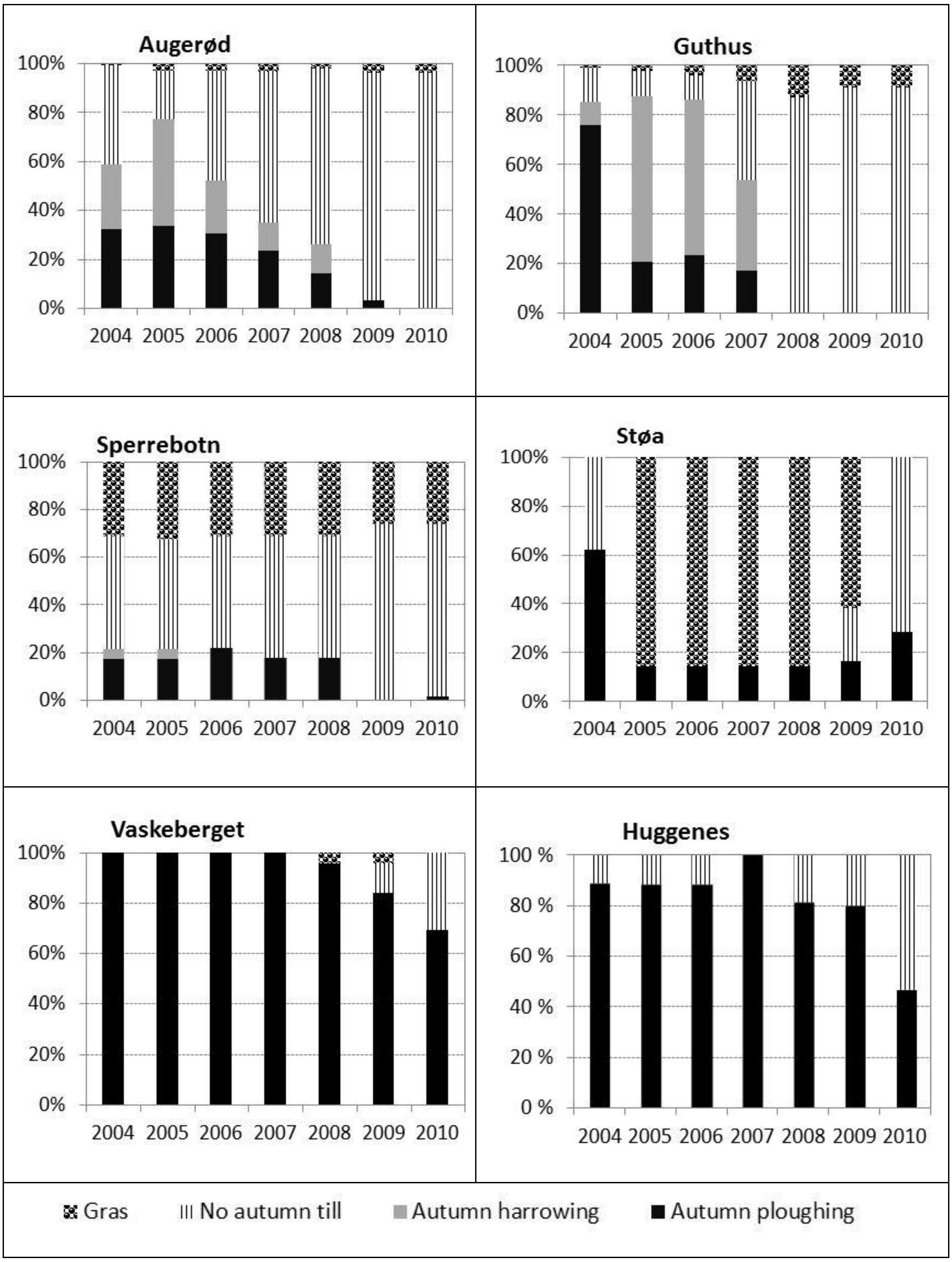


*Figure 2.* Soil management in the six catchments in the years 2004-2010

Constructed wetlands to trap SS and PP were established in the stream in Augerød in 2007/08, and in Guthus and Støa streams in 2009/10.

*Table 3.* Average* P fertilization for farms with contracts in the different catchments. nd = no data

| Catchment | 2004 | 2007 | 2008 | 2009 | 2010 |
|---|---|---|---|---|---|
| | kg P/ha | | | | |
| Augerød | 18 | 10 | 7 | 0 | 7 |
| Guthus | 21 | 19 | 10 | 2 | 2 |
| Sperrebotn | 20** | 15** | 6 | 5 | 6 |
| Støa | nd | nd | 0 | 0 | nd |
| Vaskeberget | nd | nd | nd | nd | nd |
| Huggenes | 28 | 18 | 10 | 12 | 3 |

* Part of the agricultural area included in the average values: Augerød 75 %, Guthus 100 %, Sperrebotn 92 %, Støa 70 %, Vaskeberget 0 %, Huggenes 46 %.**Estimated from Augerød and Guthus (it is expected that the farmers in Sperrebotn behaved similarly to farmers in Augerød and Guthus)

There has been a large reduction in nutrient outlet from sewage systems for private households during the last decade in the catchments. Before 2004, there was direct outlet from 1, 1 and 15 sewage systems in Støa, Vaskeberget, and Huggenes, respectively. These sewage systems were improved and the outlet reduced by 90 % in 2004/05. In 2007, 6 out of 8 sewage systems in Augerød were improved, 1 out of 3 systems were improved in Guthus and 2 remaining systems with direct outlet were renovated in Sperrebotn. Accordingly, in Augerød and Sperrebotn, reduced P concentrations during the monitoring period were expected because of the renovation of sewage systems. Sewage systems in rural areas may be serious sources of nutrients contributing to impaired water quality (e.g. WITHERS *et al.*, 2011).

***In-stream monitoring***

Mean annual runoff during the monitoring period from 2004/05 to 2010/11 was 538 mm/yr. Runoff varied from 331 mm in 2004/05 to 745 mm in 2006/07 (Table 2). All years, except 2007, experienced dry summer months with very little runoff (Figure 3).

Some summer showers may have influenced the stream flow locally, though they were not recorded by the monitoring station. During summer plant cover normally protects against soil erosion. Furthermore, evaporation is high and runoff from summer showers with intensities normal for Norway is usually low. On average, October, November and April had the highest measured monthly runoff and hence the highest transport capacity for SS and nutrients from the catchment. The traditional agricultural practice is to plough the soil to a depth of approx. 20 cm during September or October. Hence, by traditional autumn ploughing there are bare soils in the periods with the highest runoff. ØYGARDEN *et al.* (2006) showed that precipitation and runoff during the autumn months are important for the effect of tillage on SS concentrations in agricultural streams.

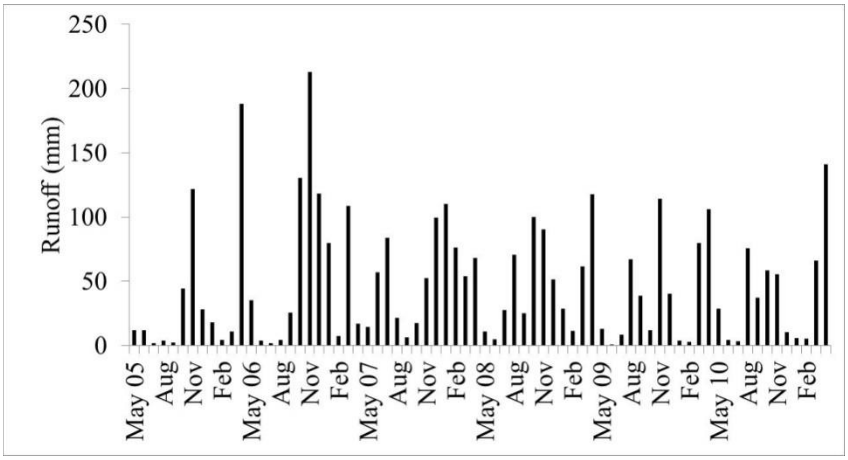


*Figure 3*. Monthly runoff (mm) during the monitoring period May 2005-April 2011
Data from Skuterud station 2005-06 and Guthus from 2006-2011

***Water quality changes***

Data from the operational monitoring of six agricultural streams in the catchment of Lake Vansjø, was analysed for potential effects of mitigation methods on water quality.

Mean TP concentrations varied from 85 µg $L^{-1}$ in the Sperrebotn stream to 146-257 µg $L^{-1}$ in the three streams from the intensive catchments with horticultural production (Table 4). As expected from earlier studies (EKHOLM *et al*., 2000) there was a relationship ($r^2$ = 0.79) between average flow normalised TP concentration for the whole monitoring period and percentage of agricultural area within the catchments. No information on forest management was available. However, TP concentrations measured in a stream in the catchment draining an area with only forest were, on average 18 µg $L^{-1}$, and the forest was anticipated to influence only slightly the variation in stream TP concentrations of the monitored streams. There was a coincidence of catchments having a high share of agricultural land and a high intensity agricultural production of potatoes and vegetables. These high intensity production systems have been shown to increase losses of P (BECHMANN et al., 2008). Comparing TP concentrations between streams may therefore reflect both production systems and share of agricultural land.

Mean TP concentrations for the six streams showed no clear trends during the monitoring period (Table 4). Losses of soil particles (SS) are an important transport process for P, and the amount of runoff is important for the losses of SS. Also, in the present study, it was found that the TP concentrations were significantly influenced by the amount of runoff ($P < 0.001$), though only 10-17 % of the variation in the streams in the cereal area was described by runoff. In the potato/vegetable area 25-45 % of the variation of TP was explained by runoff. Sampling at high flow did not always occur at the peak flow and differences in concentration at increasing and decreasing flow (hysteresis effect) may result in variation in concentrations of TP at certain flow levels (ULÉN and PERSSON, 1999). Highest concentrations of TP were measured in 2006/07 for all 6 agricultural streams. This was mainly due to high precipitation during autumn of that year and thereby high concentrations of TP, especially in November. In Støa a rain storm in August right after cutting turf grass (for sale in rolls) also resulted in very high concentration of TP.

The concentration of TP was best related to concentrations of SS for Vaskeberget and Huggenes. The relationship between TP and SS concentrations were close for most streams and most years ($r^2 > 0.6$), which indicate that erosion and soil loss is an important process for transfer of TP in the catchment. However, other sources like P fertilizer and outlet from sewage systems from private households or point sources from livestock farms in the catchments also may contribute to the variation in TP concentration.

*Table 4*. Mean annual total phosphorus (µg $L^{-1}$) and suspended sediment (mg $L^{-1}$) concentrations in the six streams

| | Mean concentrations of TP (µg $L^{-1}$) and SS (mg $L^{-1}$) | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2004/05 | | 2005/06 | | 2006/07 | | 2007/08 | | 2008/09 | | 2009/10 | | 2010/11 | |
| | TP | SS | TP | SS | TP | SS | TP | SS | TP | SS | TP | SS | TP | SS |
| Augerød | 89 | - | 110 | - | 136 | 45 | 116 | 41 | 84 | 25 | 77 | 27 | 77 | 32 |
| Guthus | 97 | - | 78 | - | 140 | 33 | 106 | 33 | 70 | 14 | 96 | 15 | 84 | 20 |
| Sperrebotn | 62 | - | 79 | - | 106 | 24 | 91 | 20 | 73 | 14 | 80 | 24 | 104 | 31 |
| Støa | 332 | - | 249 | - | 682 | 101 | 248 | 25 | 132 | 9 | 153 | 19 | 188 | 31 |
| Vaskeberget | 139 | - | 211 | - | 435 | 141 | 233 | 32 | 85 | 16 | 147 | 33 | 123 | 27 |
| Huggenes | 127 | - | 203 | - | 198 | 57 | 193 | 55 | 107 | 19 | 96 | 16 | 96 | 29 |

The dramatic changes in soil tillage practices from autumn to spring tillage in Augerød and Guthus catchments had no significant effect on concentrations of SS in autumn and winter. Since losses of soil particles and P are related, there was no significant effect of changed autumn tillage practice on concentrations of TP either. It is well known that autumn tillage results in increased concentrations of TP in runoff at the plot scale (ULÉN *et al*., 2010; LUNDEKVAM *et al*., 2002). This effect was identified for both surface and also subsurface drainage water. However, ULÉN *et al*. (2010) showed that the effect was larger at steep slopes than on flat areas. On flat areas, like in the catchment of the present study, TP losses can even be higher from areas with no autumn tillage compared to areas with autumn tillage, especially for years with low runoff (ULÉN *et al*., 2010). Furthermore, at the catchment scale, other processes, like gully erosion or landslides, may contribute and negate the effect of tillage practice on single fields (SHARPLEY *et al*., 2009). Focus on high risk areas are important for the possibility to detect significant effects of mitigation methods (HEATHWAITE *et al*., 2005), however, only focusing on high risk areas may not be sufficient to obtain the required reductions in P concentrations, especially if flat areas constitute most of the agricultural area.

The establishment of grassed buffer strips in three catchments and three constructed wetlands did not seem to reduce concentrations of TP (or SS) in the streams significantly. In Guthus though, there was a tendency to lower TP concentrations after establishment of grassed buffer strips and a constructed wetland. BRASKERUD (2002) has shown high average retention (20 - 50 %) of TP in constructed wetlands and he also showed that high loads resulted in the highest retention, because at high loads there are a higher proportion of larger aggregates which settle faster than smaller particles. The same process relates to the effect of grassed buffer strips. However, the effect of grassed buffer strips along streams in relatively flat areas is less effective than below steeper hill slopes (OWENS *et al*., 2007).

Effects of mitigation methods carried out at the field scale are generally difficult to prove at the catchment scale (HAYGARTH *et al*., 2005). Variation in weather and a limited number of samples is also a challenge to the operational monitoring. MOOSMANN *et al*. (2005) concluded that at least 30 samples were needed to detect a 3 % change in losses over 5 years. Further, JARVIE *et al*. (2010) showed that expected P losses from intensive agricultural practice could not be distinguished from P originating from small sewage systems in the rural catchments. The operational monitoring within the Vansjø area was not sufficient to detect possible effects of mitigation methods in TP and SS losses. However, analyses of data on relationship between TP and SS indicated changes as effect of reduced P application and renovation of sewage systems. Whereas reduced soil tillage in autumn, buffer strips and constructed wetlands are expected to reduce soil losses and thereby the SS and TP concentrations in the streams, reduced P application and renovation of sewage systems are expected to reduce the TP concentration relative to the SS concentration. The mean relationship of TP/SS was highest in the streams in the catchments with a high share of agriculture (Huggenes, Støa and Vaskeberget) (Table 5).

*Table 5*. Mean annual TP/SS relationship for 6 streams in the period from 2006 to 2011. Only samples with concentrations less than 150 mg SS $L^{-1}$ are included in the calculations of TP/SS.

| Year | Aug | Gut | Spe | Hug | Stø | Vas |
|---|---|---|---|---|---|---|
| | TP/SS ‰ | | | | | |
| 2006/07 | 5.9 | 7.7 | 9.1 | 10.8 | 20.6 | 15.9 |
| 2007/08 | 5.0 | 4.5 | 6.7 | 9.5 | 34.9 | 13.6 |
| 2008/09 | 3.8 | 4.0 | 5.7 | 9.2 | 20.6 | 8.1 |
| 2009/10 | 3.3 | 4.9 | 4.6 | 7.2 | 17.3 | 9.1 |
| 2010/11 | 2.6 | 3.6 | 4.9 | 5.4 | 22.9 | 9.9 |
| P-value | 0.047 | 0.001 | 0.17 | 0.28 | 0.18 | 0.24 |

Only samples with concentrations less than 150 mg SS $L^{-1}$ were included in the calculations of TP/SS, because the TP/SS relationship is also influenced by the SS concentration. By increased SS concentration the TP/SS relationship is decreased (Ref.). Omitting samples with SS concentration above 150 mg $L^{-1}$ ensure that the different years are compared within the same range of SS concentrations. All streams except Støa, showed a decreasing trend in the TP/SS relationship, but significant differences over years were only detected for Augerød and Guthus. This indicates that by the same amount of soil loss, less P were transported in the streams later years compared to the first years. In Augerød, Guthus and Sperrebotn renovation of sewage systems occurred during 2007. Therefore, in these three streams the reduction in the TP/SS relationship from 2007 to 2008 can probably partly be ascribed to this renovation. Further reduction in TP/SS may be a result of the considerable reduction in P application (60-90 %). Reduced P application will reduce the most easily releasable part of soil P, and thereby probably reduce the P release to surface runoff or to leaching through soil macro pores.

Dissolved reactive Phosphorus constitutes from 20 to 27 % of TP on average for each stream. The DRP concentrations were in all years higher in Støa than in the other streams (Figure 4). Especially in 2004/05, the mean annual DRP concentration in Støa was high, 110 µg $L^{-1}$. This year the DRP concentrations were highest in the 3 streams draining intensive potato and vegetable production areas. In the Augerød stream, the mean DRP concentrations were higher the first three years compared to the last three. Reduced P application and reduction in outlet from point sources can explain this reduction, but the reduction was not statistically significant. The number of samples analysed for DRP was relatively low in all years. In 2007/08 only two samples were taken, and therefore results from this year are not presented. Few samples make it difficult to detect any significant relationship with mitigation methods performed.

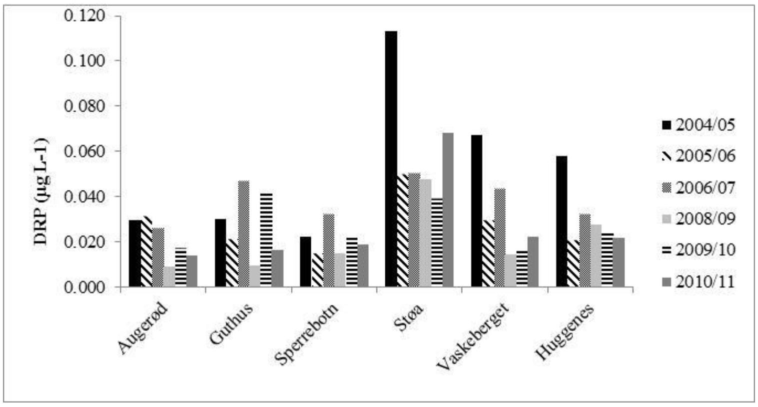


*Figure 4*. Mean annual concentrations of dissolved reactive phosphorus (DRP)

Dissolved reactive P is important for the immediate effect of P on water quality, but in the long term PP may also be available to algae (Foy, 2005). Funding of mitigation strategies are often dependant on short term results of the investment and hence mitigation methods to reduce DRP may be more important even though they may contribute less to the reduction in TP losses than methods to reduce PP losses.

**Conclusion**

Based on data from operational monitoring of six agricultural streams in a pilot area under the EU's WFD, effects of mitigation methods on TP and SS concentrations were difficult to detect over a 7 and 5 year period, respectively, despite a comprehensive implementation of mitigation methods in all sectors. The temporal variations in weather, runoff and subsequently in TP and SS concentrations are huge and a comprehensive long term monitoring program is needed to be able to show trends in concentrations even with a comprehensive implementation of mitigation methods. It was, however, possible to show a decreasing tendency of the TP/SS relationship, which could be related to both the large reduction (approx. 75 %) in P fertilizer application and renovation of sewage systems. Reduced TP/SS relationship indicates that by the same amount of soil loss, less P are transported in the streams. The concentrations of DRP decreased during the first years of the monitoring, corresponding to the renovation of sewage systems from private households, removal of pig-production in one catchment and the reduction in P fertilizer. The reductions in DRP concentrations are in the short term more important for water quality than reductions in PP, since it is regarded as being more bioavailable.

Unfortunately, milder winters and more high-intensity rainfall are expected. This will increase the risk of erosion and losses of SS and TP from agricultural areas. Thus, the effect of mitigation methods on TP losses from agricultural areas will be even more difficult to detect. However, the implementation of these methods will be more important in the future and a targeted monitoring programme must be designed to be able to detect changes in water quality.

**Acknowledgement**

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# USING GIS TO LOCATE POTENTIAL CRITICAL SOURCES OF WATER POLLUTION FROM AGRICULTURE

*Elena KONDRLOVÁ – Jaroslav ANTAL – Dušan IGAZ*

Department of Biometeorology and Hydrology, Faculty of Horticulture and Landscape Engineering, Slovak University of Agriculture, Hospodárska 7, 94976 Nitra, Slovakia, e-mail: elena.kondrlova@gmail.com; jaroslav.antal@uniag.sk; dusan.igaz@uniag.sk

**Abstract**

Soil erosion is the most important form of physical destruction of soils in Slovakia. Detached soil particles are transported and accumulated in other places down slope or they reach water bodies. The contribution of agriculture to diffuse pollution varies widely as a complex function of soil type, climate, topography, hydrology, land use and land management. Accurate spatial representation of these factors is helpful at identifying the nonpoint sources of diffuse pollution. The aim of this paper was to propose a methodology for determining the potential critical source areas of diffuse pollution of water bodies coming from agriculture in the Širočina basin (Western Slovakia) using soil loss intensity and sediment delivery calculations in GIS and data on crop rotations and nitrogen fertilizers application.

Estimation of N-fertilizer application rate came from the assumption that recommended amounts of N-fertilizers for every crop were applied in the study area to gain the possible yield. According to modelling in GIS, 309 tons of soil was transported into recipient water on the average in 2004, and 353 tons in 2005. It was estimated that 86 tons of N – fertilizers were applied to the study area both in 2004 and 2005. When looking at the parcels with the highest estimated amount of applied N – fertilizers (above 150 kg $ha^{-1}$), it can be concluded that these parcels did not have a very high potential for the pollution of water bodies. The soil loss rate calculated for these parcels was low and thus we assumed that the applied nutrients were used by plants (alfalfa) before they could be transported down slope. On the contrary, the attention of watershed managers should be focused on parcels with a high soil loss rate and high input of nutrients, e.g. parcels 12, 13, 15 in 2004 and parcels 15 and 16 in 2005. The suggested methodology presents a simple tool that allows location and enables attention to be focussed on potential sites that act as the most likely source of pollution from arable land.

**Keywords:** water erosion, diffuse pollution, nitrogen fertilizer application, GIS modelling

## Introduction

Soil, water, air, rocks and organisms are fundamental components of the biosphere. In order for soil to be able to continue to perform its irreplaceable functions, its quality and quantity must be protected. Recently, soil degradation has reached such an intensity and extent that it is considered the most serious environmental problem. In Slovakia, about 50 % of agricultural land is threatened by water erosion (ILAVSKÁ, JAMBOR, LAZÚR, 2005).

Erosion is part of the natural material cycle on the Earth. Detached soil particles from the surface layer are further transported and accumulated in lower positions or they reach water bodies. To estimate the intensity of erosion and sedimentation processes, direct and indirect measurement methods have been developed. Modelling erosion and sedimentation in geographic information systems (GIS) environment comprises a currently developing engineering tool for assessment of different variants of land use and land protection. GIS is the digital representation of geographical data that is using complex analytical tools developed in different scientific fields (HLÁSNY, 2007). Empirical and simulation GIS models progressively replace manual calculation methods. A widely used empirical model is the universal soil loss equation – USLE (WISCHMEIER, SMITH, 1978), which calculates average annual soil loss for field scale areas.

Despite the availability of many complex simulation models, USLE is still applied in Slovakia (e.g. in land consolidation projects) because of its simplicity and wide acceptance. The equation itself estimates field scale net erosion, but in combination with other procedures, it can also be used to determine the intensity of transport processes and the amount of eroded particles that can reach water bodies. For example sediment delivery ratio (SDR) can be assessed (WILLIAMS, 1977 in: JANEČEK *et al.*, 1992) based on basin area, relief ratio and infiltration rate defined by the runoff curve numbers (CN).

Soil erosion also causes direct damage to crops as well as negative changes in physical, chemical and biological properties of soil, which leads ultimately to soil fertility reduction. There is also off-site damage in other sectors, particularly in water management. Transported soil particles can clog water ways and fill up reservoirs. They can alter soils at the deposition sites by their different physical, chemical (concentration of nutrients, heavy metals, pesticide residues, etc.) and biological properties and they can cause diffuse pollution of water by substances absorbed on suspended particles or dissolved in the runoff. The negative effect on surface waters may be especially severe. In case of erosion, more complicated treatment for producing drinking water may be necessary or eutrophication (enrichment by nutrients) may be induced. This is one of the reasons why soil and water protection should be integrated (ANTAL, 2005).

Agricultural contribution to diffuse pollution varies widely as a complex function of soil type, climate, topography, hydrology, land use and land management. This complexity prevents accurate definition of contaminant sources and makes their control difficult. Thus, the contributing area of diffuse pollutants from agricultural sources depends on the coincidence of source factors (soil, crop and management) and transport factors (runoff, erosion and channel processes) (HEATHWAITE *et al.*, 2005). Knowledge about the location of these factors is helpful at identifying the nonpoint sources of diffuse pollution. Attention should be focused on high risk areas in the first instance.

The aim of this paper was to propose a methodology to determine critical source areas of diffuse pollution of water bodies coming from agriculture in the Širočina basin (Western Slovakia). Soil loss intensity and sediment delivery in GIS were calculated and data on crop rotations and nitrogen fertilizers application were used.

## Materials and methods

### *Study area*

The Širočina basin covers an area of 106 km$^2$ and it is one of the sub-basins of the Žitava river that is one of the main tributaries of the Nitra River in the western part of Slovakia. The basin is located in the Nitra district between the municipalities of Zlaté Moravce on the north and Vráble on the south. The average annual air temperature is 10 °C and the average annual precipitation reaches 590 mm. It belongs to the corn and sugar beet production region with a climate moderately warm to warm. The land use structure is 62.5 % agricultural land, 30.5 % forests, other wooden vegetation 3.2 %, built-up area 5.2 %, pastures 2 %, vineyards 1 %. There are three small (3 up to 17 ha) multi-purpose water reservoirs in the basin. Although they were used for irrigation in the past, nowadays they serve fishery, recreation and flood protection. Because of the lack of detailed data on crop rotation in the whole basin, a study site was selected for data collection. Detailed information on crop rotation in the years 2004 and 2005 was provided by Agro NV, Ltd. Nemčiňany for 20 parcels located in the central part of the basin (Figure 1). The study area is divided into two production regions: corn production region (CPR) on the south (parcel 15 to 20) and sugar beet production region (SBPR) on the north (parcel 1 to 14).

***Modelling in GIS***

After obtaining all necessary data, input layers for modelling were prepared with ArcMap v.10 (ESRI). The average annual soil loss on agricultural land in the Širočina catchment was determined using USLE (WISCHMEIER and SMITH, 1978):

$$A = R * K * L * S * C * P \qquad (1)$$

where A – average annual soil loss ($t\ ha^{-1}\ y^{-1}$)

R – rainfall erosivity factor ($MJ\ mm\ ha^{-1}\ h^{-1}\ y^{-1}$)

K – soil erodibility factor ($t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1}$)

L – slope length factor (no dimension)

S – slope gradient factor (no dimension)

C – factor of protective impact of vegetation cover (no dimension)

P – factor of soil protection measures (no dimension)

Slope length factor and slope gradient factor were substituted by the combined topographic factor LS (WISCHMEIER, SMITH, 1965):

$$LS = l_d^{0.5} * (0.0138 + 0.0097 * s + 0.00138 * s^2) \qquad (2)$$

where: $l_d$ – continuous slope length (m)

s – slope gradient (%)

Not all eroded soil particles are transported into water courses: some of them could be deposited down slope. To estimate sediment discharge from the basin, USLE calculation was reduced with sediment delivery ratio. We used the approximate method by WILLIAMS (1977):

$$SDR = 1.366 * 10^{-11} * P_p^{-0.0998} * S_r^{0.3629} * CN^{5.447} \qquad (3)$$

where: $P_p$ – basin area ($km^2$)

$S_r$ – relief ratio ($m\ km^{-1}$)

CN – runoff curve number (no dimension)

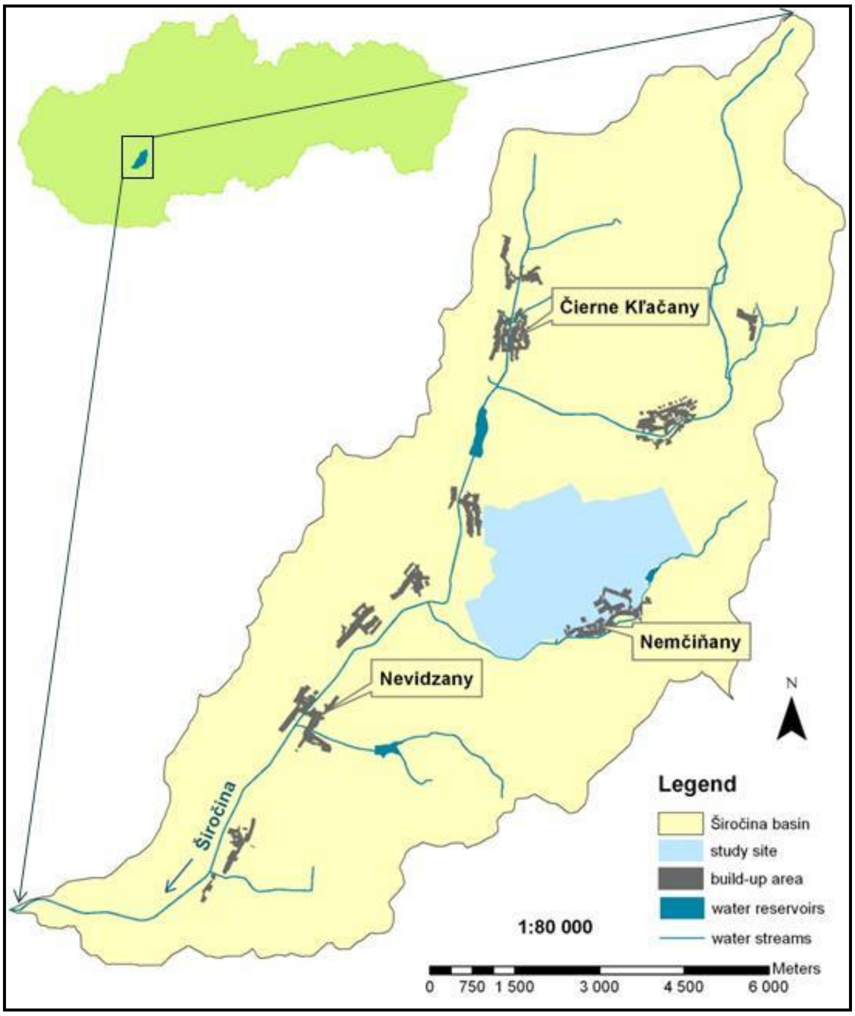


*Figure 1.* Position of the Široćina basin and the study area in Slovakia

CN was determined on the basis of land use, hydrologic condition and hydrologic soil group (ANTAL, 1997). To include the impact of retention components and their spatial representation, averaged CN method was used (ŠINKA, 2008; ŠINKA, 2009), where CN value of every grid cell was dependant on CN values of grid cells in its neighbourhood.

Estimation of N-fertilizer application rate was based on the assumption that the recommended amount of N-fertilizers was applied for every crop. No. 199/2008 Coll. Regulation includes recommended amount of N-fertilizer for every crop (Table 1). For our purposes, it was sufficient to use the mean hectare crop yields published by the Research Institute of Agriculture and Food Economics (Table 1). The estimated amount of applied N-fertilizers was calculated using following formula:

$$N_{est} = N_C * Y_M \tag{4}$$

where: $N_{est}$ – estimated amount of applied N-fertilizers for specific crop (kg ha$^{-1}$)

$N_C$ – need of nitrogen of every specific crop in rotation (kg t$^{-1}$)

$Y_M$ – average yield of specific crop (t ha$^{-1}$)

*Table 1*. Overview on crop rotations, crop need of nitrogen (N) and gained mean crop yield in 2004 and 2005

| Crop | Parcel | | Need of N | Mean yield in 2004 (t ha$^{-1}$) | | Mean yield in 2005 (t ha$^{-1}$) | |
|---|---|---|---|---|---|---|---|
| | 2004 | 2005 | (kg t$^{-1}$) | CPR | SBPR | CPR | SBPR |
| Alfalfa | 5,6,7,11,14 | 5,6,11,14 | 6.00 | 32.90 | 26.50 | 28.94 | 28.25 |
| Corn | ---- | 8 | 27.00 | 6.87 | 7.23 | 8.39 | 6.45 |
| Hybrid rye | 10 | 20 | 23.00 | 4.63 | 3.68 | 3.36 | 3.82 |
| Oilseed rape | ---- | 10,16,17,19 | 45.00 | 3.09 | 3.27 | 2.27 | 2.85 |
| Silage corn | 15 | ---- | 3.00 | 30.45 | 33.01 | 38.56 | 31.80 |
| Spring barley | 4,19,20 | 7,17,18 | 23.00 | 5.04 | 4.61 | 4.39 | 3.92 |
| Sunflower | 12,13 | 1,2,9 | 55.00 | 2.55 | 2.32 | 2.44 | 2.26 |
| Winter rye | 1,2 | 3,15 | 23.00 | 4.63 | 3.68 | 3.36 | 3.82 |
| Winter | 3,8,9,16,17,18 | 4,12,13 | 23.00 | 5.54 | 5.60 | 4.85 | 5.12 |

**Results and discussion**

The computation of average soil loss and sediment delivery ratio was processed in the GIS interface using the individual input layers that were combined in Map Algebra tool – Raster Calculator. Soil loss into water streams and into the three small water reservoirs in the Širočina basin was determined (KONDRLOVÁ, 2009). Figure 2 shows the basic steps to localise the sources of diffuse pollution. The calculated values of mean soil loss for every grid cell are shown in Figure 3 (year 2004) and Figure 4 (year 2005), respectively.

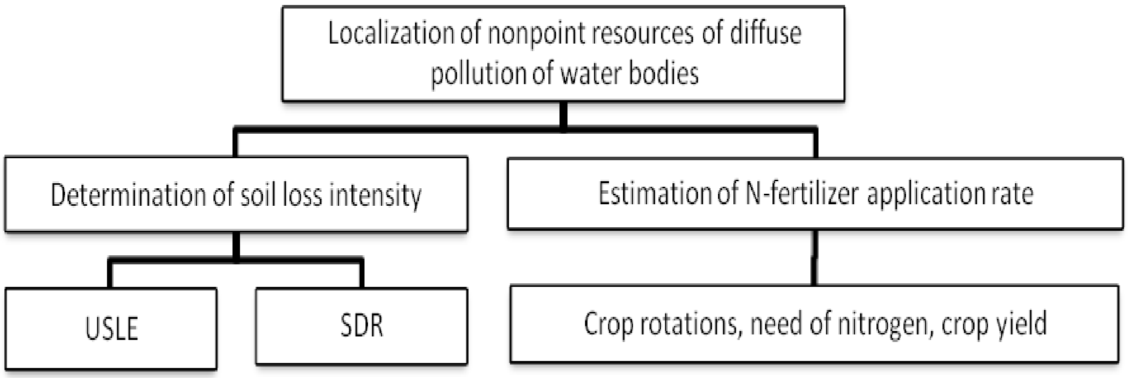


*Figure 2.* Scheme showing steps of the proposed methodology

Reclassifying the estimated raster into 5 classes helped to visualize the areas with the highest calculated mean soil loss (above 7 t ha$^{-1}$ y$^{-1}$). According to our calculations, parcels 12, 13 and 15 were the most threatened by water erosion in 2004, and parcels 15 and 16 in 2005.

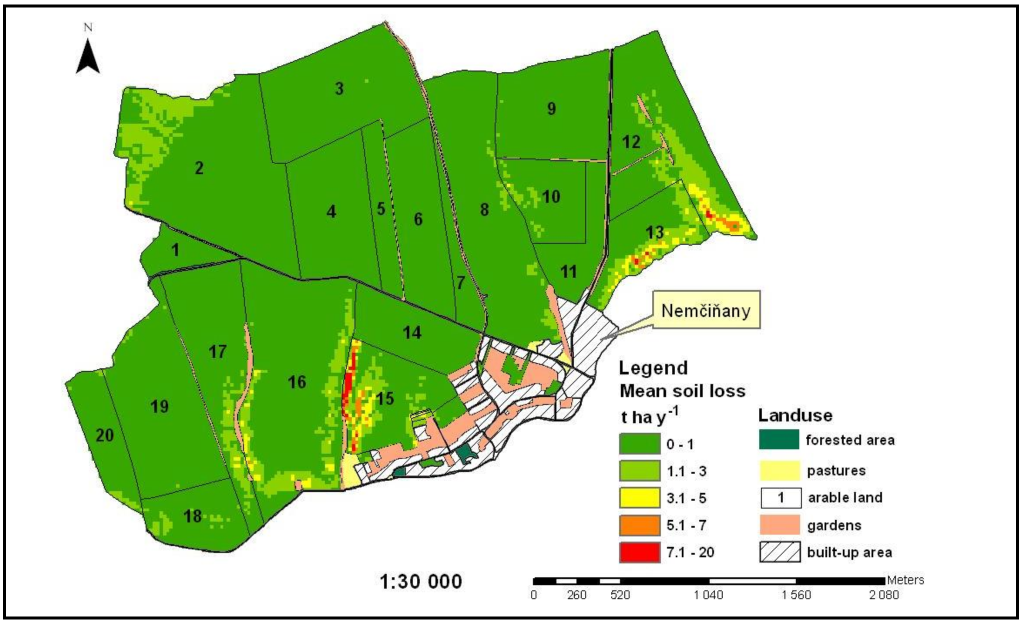


*Figure 3*. Calculated water erosion intensity in 2004

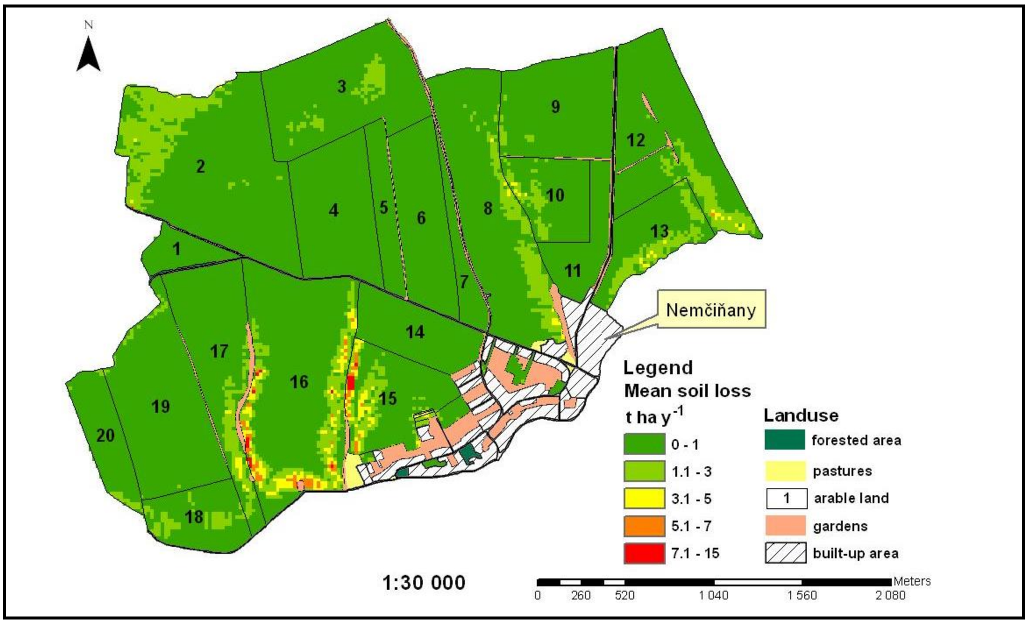


*Figure 4*. Calculated water erosion intensity in 2005

Using another tool of Spatial Analyst extension (Zonal Statistics as a Table), the mean soil loss rate for every polygon parcel in the study site was calculated as weighted and arithmetic mean from grid cell values (Table 2).

The mean soil loss rate reached 1.59 t ha$^{-1}$ in 2004 and 1.18 t ha$^{-1}$ in 2005, respectively. The highest values can be seen in Figures 3 and 4. To determine total soil loss, mean soil loss rate was multiplied by the parcel area. According to modelling in GIS, 309 tons of soil was transported into the recipient water bodies in 2004, and 353 tons in 2005.

*Table 2.* Estimated mean soil loss and amount of applied N-fertilizers in the study area

| Parcel | Area (ha) | 2004 Mean soil loss rate (t ha$^{-1}$ y$^{-1}$) | 2004 Soil loss (t) | 2004 $N_{est}$ (kg ha$^{-1}$) | 2004 $N_{est}$ * area (t) | 2005 Mean soil loss rate (t ha$^{-1}$ y$^{-1}$) | 2005 Soil loss (t) | 2005 $N_{est}$ (kg ha$^{-1}$) | 2005 $N_{est}$ * area (t) |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 8.87 | 0.1203 | 1.07 | 84.64 | 0.75 | 0.1557 | 1.38 | 124.30 | 1.10 |
| 2 | 85.10 | 0.5311 | 45.20 | 84.64 | 7.20 | 0.6873 | 58.49 | 124.30 | 10.58 |
| 3 | 51.00 | 0.3809 | 19.42 | 128.80 | 6.57 | 0.5392 | 27.50 | 87.86 | 4.48 |
| 4 | 37.58 | 0.2316 | 8.71 | 106.03 | 3.98 | 0.1856 | 6.98 | 117.76 | 4.43 |
| 5 | 12.83 | 0.0213 | 0.27 | **159.00** | 2.04 | 0.0210 | 0.27 | **169.50** | 2.17 |
| 6 | 31.87 | 0.0188 | 0.60 | **159.00** | 5.07 | 0.0188 | 0.60 | **169.50** | 5.40 |
| 7 | 9.19 | 0.0047 | 0.04 | **159.00** | 1.46 | 0.0339 | 0.31 | 100.97 | 0.93 |
| 8 | 61.49 | 0.2880 | 17.71 | 128.80 | 7.92 | 0.5760 | 35.41 | 174.15 | 10.71 |
| 9 | 38.27 | 0.1869 | 7.15 | 128.80 | 4.93 | 0.3424 | 13.10 | 124.30 | 4.76 |
| 10 | 21.15 | 0.4390 | 9.28 | 84.64 | 1.79 | 0.5684 | 12.02 | 128.25 | 2.71 |
| 11 | 15.84 | 0.0388 | 0.61 | **159.00** | 2.52 | 0.0393 | 0.62 | **169.50** | 2.69 |
| 12 | 53.39 | **0.8443** | 45.08 | 127.60 | 6.81 | 0.4605 | 24.59 | 117.76 | 6.29 |
| 13 | 22.95 | **1.3374** | 30.69 | 127.60 | 2.93 | 0.7295 | 16.74 | 117.76 | 2.70 |
| 14 | 22.86 | 0.0325 | 0.74 | **159.00** | 3.63 | 0.0329 | 0.75 | **169.50** | 3.87 |
| 15 | 29.44 | **1.5926** | 46.89 | 91.35 | 2.69 | **1.1772** | 34.66 | 77.28 | 2.28 |
| 16 | 79.55 | 0.5147 | 40.95 | 127.42 | 10.14 | **0.9431** | 75.02 | 102.15 | 8.13 |
| 17 | 40.03 | 0.3477 | 13.92 | 127.42 | 5.10 | 0.4347 | 17.40 | 100.97 | 4.04 |
| 18 | 19.95 | 0.4676 | 9.33 | 127.42 | 2.54 | 0.5846 | 11.66 | 100.97 | 2.01 |
| 19 | 51.59 | 0.1477 | 7.62 | 115.92 | 5.98 | 0.2162 | 11.16 | 102.15 | 5.27 |
| 20 | 17.32 | 0.2168 | 3.75 | 115.92 | 2.01 | 0.2457 | 4.26 | 77.28 | 1.34 |
| **Σ** | 710.26 | ---- | 309.03 | --- | 86.06 | ---- | 352.92 | ---- | 85.88 |

Based on N-fertilizer recommendation, a simple tool was proposed to determine the quality of transported particles. Table 2 and Figures 5 and 6 show the estimated amount of applied N – fertilizers on individual parcels in the study area in 2004 and 2005. The values depended on the N – nutrient requirement of crops. More than 150 kg ha$^{-1}$ of N – fertilizers was recommended for parcels 5, 6, 7, 11 and 14 in 2004, and for parcels 5, 6, 8, 11 and 14 in 2005. Parcels 3, 8, 9, 12, 13, 16, 17 and 18 were fertilized with a N amount between 125 and 150 kg ha$^{-1}$ in 2004. In 2005, this amount was recommended for parcel 10. Based on areas of parcels and applied N-fertilizer rates, it was estimated that 86 tons of N – fertilizers was used both in 2004 and 2005.

Comparing the graphical (Figures 3 – 6) and numerical (Table 2) results helped to localize critical areas that could be potential sources of diffuse pollution. Interestingly, parcels with the highest estimated amount of applied N – fertilizers (above 150 kg ha$^{-1}$) did not have a very high potential for pollution of water bodies. The soil loss rate from these parcels was low in both years so the applied nutrients could be used by plants (alfalfa) before they were lost to erosion. In contrast, parcels with a high soil loss rate and a high input of nutrients, e.g. parcels 12, 13, 15 in 2004 (sunflower and silage corn), and parcels 15 and 16 in 2005 (winter rye and oilseed rape) should be in the focus of watershed management. Primary method to reduce water pollution should be the decrease of soil loss by soil protection measures. An alternative way could be to take into consideration crop rotations not only because of the protective impact of vegetation cover, but also because of the different needs of nutrients.

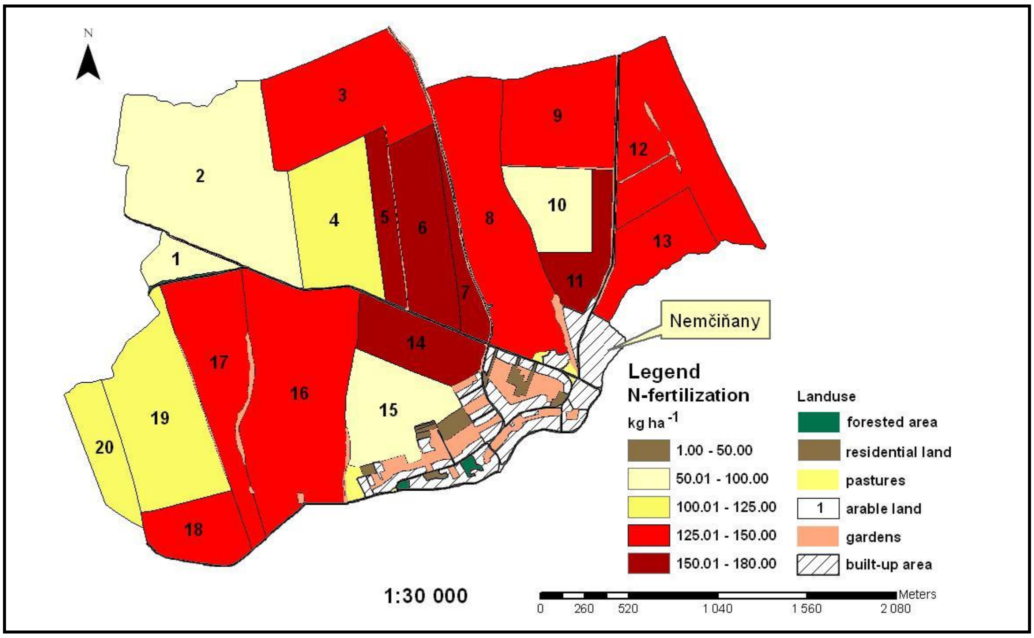


*Figure 5.* Estimated amount of applied N-fertilizers in 2004

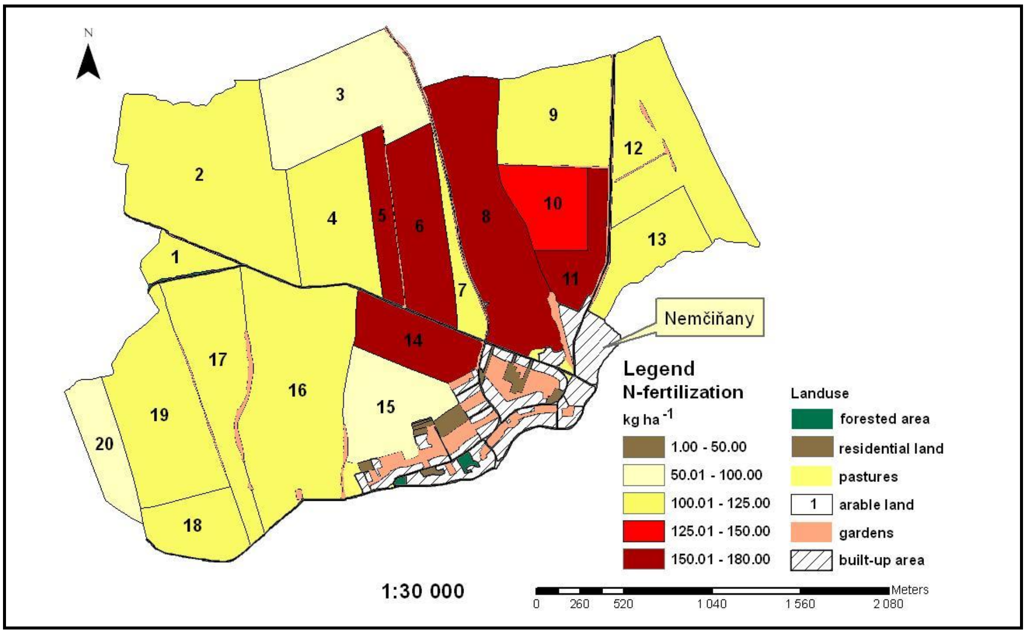


*Figure 6.* Estimated amount of applied N-fertilizers in 2005

### Conclusions

Soil erosion intensity is nowadays calculated in GIS for several the purposes (e.g. land consolidation, landscape planning, water management). With our proposed method, potential areas for the pollution of water bodies can be determined. This approach focuses on potential sites that can be the most likely sources of pollution from arable land. It should be noted that even with high quality input data available, a modelled situation provides only a general guidance for river basin managers on sediment quality and quantity due to water erosion.

**Acknowledgements**

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# RILLS SWITCH CATCHMENTS INTO A HIGHER PHOSPHORUS-LOAD MODE - A CASE STUDY

*István SISÁK – Tamás PŐCZE – Péter SZŰCS – Csaba HAUSNER – Ferenc MÁTÉ*

Department of Plant Production and Soil Science, Georgikon Faculty, University of Pannonia, 16 Deák F. st., H-8360 Keszthely, Hungary, talajtan@georgikon.hu

**Abstract**

The water quality of Lake Balaton had improved following various interventions which reduced the external phosphorus load between the late 1970`s and early 1990`s. However, improvement seems to have been reversed in recent years and hypertrophic conditions can be frequently observed during the summer in the shallow water near the shoreline. The bad water quality is also not uncommon in the open water in the Keszthely- and Szigliget-basin. Further reduction in the external phosphorus (P) load must be achieved to reach targets set by the Water Framework Directive (WFD). The loess derived and eroded soils in the southern sub-watershed of the lake are largely under agricultural cultivation. The Somogybabod study catchment (7 km$^2$) belongs to the catchment of the Tetves stream that drains an area of approximately 80 km$^2$ into the lake. Only 35 % of the study catchment is arable land, the rest is woodland, orchards and shrubs along the ephemeral stream in the valley bottom. Rainfall and water level of the stream were continuously monitored at the outlet of the catchment in 2006. Samples were automatically taken during runoff events and discharge, suspended sediment load and P loss were calculated. Three periods of heavy rainfall in June 2006 produced excessive runoffs and soil losses and carved several rills into the soil of the arable fields. 19 % of the catchment is affected by rill erosion. The lengths of rills were derived from satellite images and the calculation was controlled by GPS measurements. The rills were monitored on the field for cross section area and total soil loss from rills was calculated by using soil bulk density measurements. Digital elevation models (DEM) were produced with superimposed random roughness and with incised rills. Flow lengths have been decreased and slope angles have been increased approximately by a ratio of ten percent due to the rills for five erosion areas on arable land. Aerial photos were used to digitize strongly eroded areas on arable land. The calculated soil loss removed from rills (1227 tons) was comparable with the measured sediment loss at the outlet (1372 tons). Phosphorus enrichment is only 104–108 % compared to the original soil in the major runoff events thus, bulk soil was eroded en-masse when most of the suspended sediment passed the outlet and P-enriched soil was transported only in the latter phase of the runoff. Partial coincidence of strongly eroded areas, rill heads and convex slopes indicate that periodical rill formations significantly contribute to the erosion and P load from the catchment and steep slopes in shoulder position produce more runoff and soil loss after rill formation in this transport capacity limited erosion environment. Our conclusion is that prevention of rill formation must be a priority tool to reduce P load into Lake Balaton and to achieve WFD target.

**Keywords:** rill erosion, loess derived soils, Lake Balaton, water quality, Water Framework Directive, phosphorus load

## Introduction

Lake Balaton, the largest shallow lake in the Central European region, is sensitive to eutrophication. The water quality had rapidly deteriorated since the early 1970s (SOMLYÓDY, 1983). However, both total P and Chlorophylla (Chl-a) content in Lake Balaton decreased significantly in the last decade due to the reduced external loads (SOMLYÓDY *et al*., 2003). The control of point sources (wastewater treatment around the lake) lead to the current situation that the major portion of phosphorus emissions is associated with diffuse sources. Since the current P load is about two times higher than the desirable value, future water quality management of the lake must reduce the non-point pollution (SOMLYÓDY & HOCK, 2002, KOVÁCS *et al.*, 2008). In spite of all the efforts and achievements in reducing phosphorus loads to the lake, large parts of the water body are still eutrophic in the most sensitive tourist season (KVVM, 2011). Water Framework Directive (WFD) of the EU (EC, 2000) requires member states to implement River Basin Management Plans to achieve good surface water quality status by 2015. However, just as in many other countries, Lake Balaton and Hungary are unlikely to meet these requirements by 2015.

Loess derived silty soils are generally prone to erosion (RENARD et al., 1997) and strong rill formation and disastrous events may occur on loess areas (BRYAN 2000, VAN DIJK *et al*. 2005). These kinds of parent materials are common in Hungary (21.8 %) but they are over proportional in the watershed of Lake Balaton (32 %) and for our study catchment at Somogybabod (32.2 %). Loess-like wind blown deposits for Hungary areas shown on Figure 1.

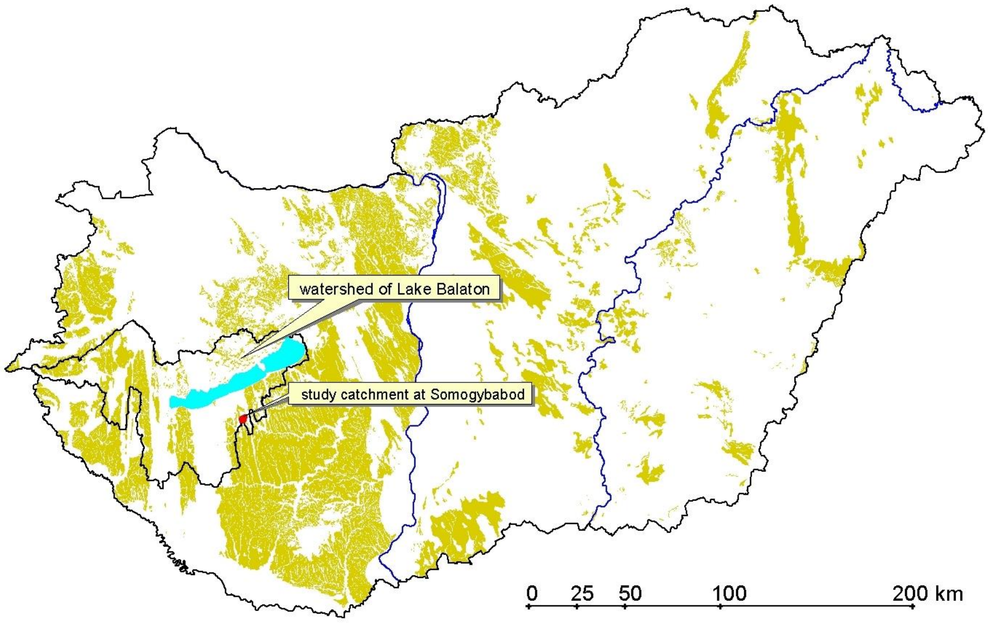


*Figure 1*. Loess-like wind blown deposits in Hungary (PELIKÁN & PEREGI, 2005)

Rill erosion is an accelerated form of soil erosion by water which results in much larger amount of soil loss than sole rain splash or sheet wash erosion (MEYER, 1975, MORGAN, 2005). The off-site effects of soil erosion are most often connected to rill and gully erosion (LAL, 1998) which may sometimes worsen into disasters (VAN DIJK *et al*., 2005). That is why VANDEKERCKHOVE *et al*. (2004) have suggested catchment scale monitoring of rills as an indicator and a potential policy tool in a preparatory study for the EU Thematic Strategy for Soil Protection.

Our objectives were to relate catchment-scale monitoring of rills to the directly measured soil loss at the outlet of our study catchment and to draw conclusions on the acceleration of erosion processes and total phosphorus load by riling.

## Materials and methods

### *Study area*

Lake Balaton is located in the western part of Hungary, in Transdanubia, between northern latitudes 46°62' and 47°04', and eastern longitudes 17°15' and 18°10' (Figure 1). The total area of the lake itself plus its watershed is 5775 $km^2$. The soil forming factors show considerable variability around Lake Balaton. Pleistocene loess and fluvial sand overlay are common in the southern part where our study catchment is located. Eutric Luvisols formed on loamy sand and finer deposits are the major soil types under cultivation in the south (SISÁK & MÁTÉ, 2003).

The study catchment at Somogybabod is 7.03 $km^2$ and is located in the catchment of the Tetves-stream which drains an area of approximately 80 $km^2$ into the lake. The study catchment consists of limited number of land units, thus causalities between agricultural activities and erosion and P loss could be revealed and USLE and other erosion models can be successfully applied. On the other hand, it is large enough to be considered as hydrologic units in a large hierarchical sub-watershed-watershed system. Catchments with a few square kilometre area have the proper size to represent interface between agriculture and water management, therefore they can be the proper demonstration tools to communicate research results both to agricultural and watershed managers.

The study catchment is covered with forest (56.8 %) and orchards (4.8 %) in the upper segment, arable land (35.2 %) is located in the lower segment and bushes and woodlands (3.2 %) are along the ephemeral stream in the valley. Arable farming in the area includes all the usual cash crops. Even maize and sunflower are not exceptions in spite of their low surface cover factor and high erosion risk. The lowest and highest elevations are 155 m and 271 m above the Adriatic Sea and the average slope is 9.9 %. The steepest slopes are usually forested. There are places even on arable land with gentler slopes where the calcareous sandy loess parent material is on the surface so that the whole soil solum is eroded. Rill and gully erosion are common in the catchment and in the whole region (Figure 2).


*Figure 2*. The study catchment at Somogybabod
(green: forest, orchards, bushes; white: arable land; yellow: areas eroded down to the calcareous parent material; red lines: rills shown only within delineated fields ; 1..5: delineated areas of 11.1-31.9 ha drained by different groups of rills)

***Measurements with the monitoring station***

A monitoring station (SISÁK & MÁTÉ, 2003) was installed at the outlet of the catchment and it was equipped with automatic American Sigma 950 sampler, ultrasonic water level recorder and meteorological extensions (rainfall, temperature and relative humidity measurements). The system is able to maintain the standby mode for one week and takes 24 samples triggered by the increase of water level above a pre-set value. Operation is usually stopped in winter between December and March. Data generated in June 2006 is presented here.

The station was built with a Parshall flume. Water level and discharge relationship for the flume is the following:

$$Q = 2.367 * b * h^{1.566}$$

where $Q$ is discharge in $m^3s^{-1}$; $h$ is head in centimetres and $b$ is throat width of the flume in meter.

The sampler takes 24 composite samples from the bottom flow composed of three subsamples at equal time intervals. The sampling started when water level exceeded a depth of 8 cm. Short intervals (5 minutes) were set in summer when heavy rainstorms may occur. Sample size was 0.3 litres. Samples were collected either on the day of sampling or on the following day, delivered to the laboratory and analysed immediately or on the next day. Suspended sediment content of the samples was determined by filtering the thoroughly shaken suspension through a membrane with 0.45 µm pore diameter and by drying the filtrate at 105 °C. Total P (TP) content was determined on unfiltered samples following digestion with a semi micro Kjeldahl procedure (BREMNER & MULVANEY, 1982) by the colorimetric method of MURPHY & RILEY (1962). Soil loss and P loads were calculated by multiplying concentrations with discharge of the respective period.

***Digital data sources and their processing***

Rills were detected on and digitized from a SPOT space image taken on 15th July 2006 published by Google Earth. The image section depicting Somogybabod and its surroundings was geo-referenced based on the 1:10,000 digital topographic map and rills were digitized on the screen. The eroded spots on the arable land within the catchment were digitized according to the whitish spots on a geo-referenced air photo taken in 2000. Digital elevation models (DEM: pixel sizes of 10 and 0.5 metre) of the area were generated from the contour lines of the 1:10,000 topographic maps (FÖMI, 2012). Random roughness was superimposed to the DEM of 0.5 m with maximum vertical deviation of ± 20 cm (average deviation ± 10 cm) and digitized rill lines were incised (one pixel wide 40 cm deep). Local depressions were eliminated by fill procedure after each step. 10 metre DEM was used to delineate different curvature categories and 0.5 metre DEM was used to calculate flow lengths. ArcMap program was used (ESRI, 2012) for the GIS procedures described in this paragraph.

***Rill and soil monitoring***

6.4 % of the rills were monitored for their geometry (width, depth, cross section shape, length) and soil samples were collected to determine bulk density of the soil within and below the rill depth. The monitored rills were recorded with GPS and the results were compared with the previously digitized rills to check validity of digitization. The average results of the monitored rills were used to assess soil loss from all rills. Arable soils were also monitored for their average total phosphorus content (WITHERS *et al*., 2007).

**Results and discussion**

Three major runoff events were monitored in June 2006 with peak discharge rates around 500 $m^3*(5\ min)^{-1}$. produced by 18–26 mm rainfalls (Figure 3). Two events directly followed each other on the same day (29th June) in midday and in the evening. The former was triggered by a much larger rainfall (25.91 mm) than the latter (17.53 mm) but the latter resulted in much larger discharge and soil and phosphorus losses (Table 2). The runoff event on 14th June was preceded by smaller rainfalls. The three events generated huge soil (1972 tons) and phosphorus (878 kg) loads at the outlet. These three events gave the bulk of the yearly load in 2006. Dominating events in summer is the usual erosion pattern in Hungary (HORVÁTH *et al.*, 2012) and elsewhere in the world with a continental climate (RENARD *et al.*, 1997).

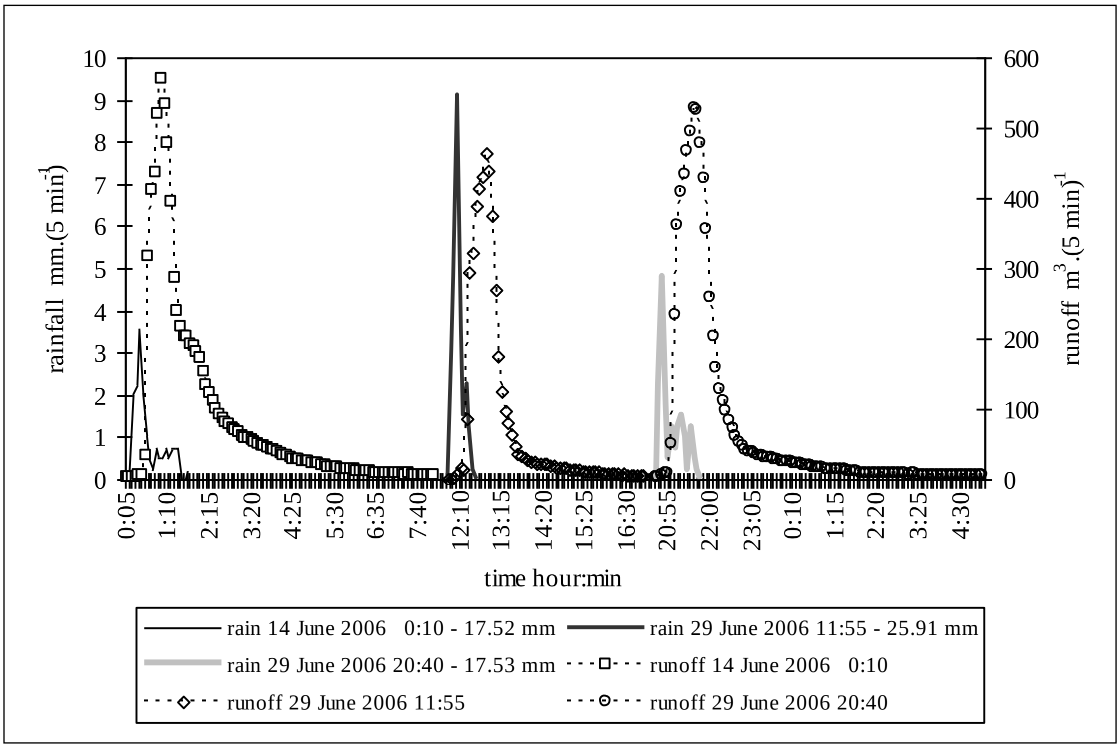


*Figure 3*. Three major run-off events in June 2006

*Table 2*. Measured and calculated properties of three major runoff events at the outlet of the research catchment at Somogybabod

| Event | Date and time | Rainfall (mm) | Runoff ($m^3$) | Soil loss (T) | Total P loss (kg) |
|---|---|---|---|---|---|
| 1 | 14 June 2006 0:10 | 17.52 | 8105.7 | 520.0 | 332.8 |
| 2 | 29 June 2006 11:55 | 25.91 | 4708.0 | 327.1 | 214.4 |
| 3 | 29 June 2006 20:40 | 17.53 | 7125.8 | 525.1 | 330.7 |
| Total | | 60.96 | 19 939.5 | 1372.2 | 877.9 |

The runoff rates were rather high for event 1 and 3 (6.61 and 5.81 % of the total rainfall amount respectively) and much less for event 2 (2.6 %). The events occurred in a very short time interval thus surface cover must have been very similar therefore differences in runoff and erosion are due to the variance in rainfall intensity, antecedent soil moisture content and presence or absence of rills. The total P content of the original soil was

605 mg.kg$^{-1}$ (WITHERS *et al.*, 2007). The total P contents of suspended sediment in the monitored major runoff events were 630–655 mg.kg$^{-1}$ which means P enrichment varied between 1.041–1.083 (1.058 on the average). The low P enrichment shows that bulk soil was eroded en-masse when most of the suspended sediment passed the outlet and P-enriched soil must have been transported only in the late receding phase of the runoff.

A weak form of the desertification, with an increase in aridity can be detected in Hungary (EEA 2010). At the same time, the frequency of disastrous flash floods has significantly increased in the last decades (BARTHOLY & PONGRÁCZ, 2007). Dry areas of the watershed of Lake Balaton with loess derived soils show serious Mediterranean type erosion with rills and gullies (GÁBRIS *et al*., 2003, JAKAB, 2008) which may easily lead to a badland–scenario if increase of aridity and flash flood frequency continue. This process forecasts an increased P load for the lake in the future (SISÁK *et al*., 2008).

The rill monitoring brought relatively consistent results. Dry bulk density of the plough layer (usually equal to the rill depth) was 1.29 kg.dm$^{-3}$ and it was 1.54 kg.dm$^{-3}$ for the rill bottom with small variability independent on weather they consisted of non-calcareous soil solum material or calcareous parent material (Table 3). The total length of the monitored rills were 655 meter, 6.4 % of the total rill length (10175 m) which was derived from the satellite image. GPS measurements of rills largely coincided with rills extracted from the image. 120.6 kg soil was lost from one meter rill on the average and this value was used to calculate total soil amount removed by runoff from the rills (1226.8 tons). That is 89.4 % of the total soil loss measured at the outlet in the three major events. The contribution of gully erosion to the sediment production from cultivated catchments may reach 44-80 % of the total loss (POESEN *et al*., 1996). Our result shows that the ratio of rill erosion may be even higher for short periods with heavy rainfall.

Rills could be detected only on arable land. Arable land amounts 35.2 % of the catchment area while the rills drain only 19 % of the total area thus, little more than half of the arable land is prone to rill erosion. Unfortunately, most of these rills are directly connected with the ephemeral stream at the valley bottom (Figure 2). Figure 2 shows only those rills (approximately 90 % of the total rill length) which are within the delineated fields.

*Table 3*. Average and standard deviation of bulk density and calcium carbonate content for the soil samples along rills n=19

| Samples | Dry bulk density kg.dm$^{-3}$ mean (st.d.) | Calcium-carbonate content % mean (st.d.) |
|---|---|---|
| Plough layer | 1.29 (0.05) | 3.72 (4.3) |
| Rill bottom | 1.54 (0.06) | 3.71 (4.2) |
| Sediment deposition | 1.36 (one sample) | 2.29 (one sample) |

Total soil and phosphorus losses given in Table 2 were calculated for different source areas (Table 4).A relatively large area is covered by forest and the steepest slopes (above 25 % slope gradient) are all forested. Certainly, these areas are prone to erosion in spite of the dense cover. But we can certainly assume that most of the soil loss comes from arable land with rills. Therefore, the local erosion rates may vary between almost zero and approximately 36 t.ha$^{-1}$.yr$^{-1}$ by which the latter corresponds with 2.77 mm.yr$^{-1}$ soil loss. Even larger erosion rate estimates are realistic locally. Large amount of soil must have come from the most eroded spots where the parent material is on the surface and 1.0–1.4 metre soil solum has been eroded since the early 19$^{th}$ century the beginning of the cultivation. For simplicity, we calculated with 200 years and 1000-1400 mm soil solum thus, the erosion rate became 5-7 mm per year (65–90 t.ha$^{-1}$.yr$^{-1}$). Considering the fact that weather conditions may be excessively variable year by year, erosion on certain spots in rainy years may reach several hundred tons per hectare. This high rate of erosion is only possible if riling has been widespread and common in the catchment for the last two centuries and the erosion by cultivation was strong.

*Table 4*. Average soil and total phosphorus loss in June 2006 from the research catchment at Somogybabod considering different contributing areas

| Considered contributing areas | Area | Soil loss | Total P loss |
|---|---|---|---|
| | ha | $t.ha^{-1}$ | $kg.ha^{-1}$ |
| total area | 703 | 1.95 | 1.25 |
| agricultural area | 281 | 4.88 | 3.12 |
| arable land | 245 | 5.55 | 3.55 |
| arable land affected by rill erosion | 134 | 10.28 | 6.58 |
| strongly eroded spots | 39 | 35.64 | 22.74 |

Five arable land sections were delineated within a very large field where the sections were drained by separate groups of rills. These erosion fields varied in size between 11.1 and 31.9 hectares with an average of 19.2 hectares (Figure 2). The highly eroded spots within the five areas amount to 15.6 %. The distribution of curvature values (Figure 4) within the total area of the five fields approaches symmetrical distribution with highest frequency at straight slopes (42.7 %) and lower frequencies at convex and concave slopes (28.5 and 28.8 % respectively). However, the distribution for the highly eroded spots shows the strong presence of convex slopes (35.1 %) to the detriment of the concave slopes (22.5 %) with the unchanged frequency of the straight slopes (42.4 %) compared to the whole area. Figure 2 also shows that rills very often start in highly eroded spots or at the shoulder or backslope position with convex or straight slopes.

Sharpley *et al*. (1998) have shown that high percentage of the diffuse phosphorus pollution may originate from a relatively small part of the watersheds and the majority of the pollution is in particulate form for the intensively used agricultural land. Landscape features can be associated with accelerated erosion. Pennock & de Jong (1987) have found that convergent shoulders and divergent backslopes, where curvature and/or slope values were ranked first among the landform elements with the highest erosion which is also confirmed by our study. The dominance of transport capacity limited erosion (Morgan, 2005) in this environment also reinforces the importance of the upslope areas as sources of soil loss.

A small GIS study was also conducted to find out the influencing factors of higher erosion in presence of surface random roughness and rills. The longest flow path within five delineated fields with smooth digital elevation model was 772 m on the average while it was 1318 m with superimposed random roughness. Random roughness almost doubled the flow length hence halved the real slope angle. The effect of incised rills was the opposite although much smaller in size (Table 5). They reduced the flow length and increased the average slope angle by approximately one tenth of the original value.

The importance of the surface roughness in the concentrated surface runoff generation and erosion has been explored by many authors (Kirkby *et al*., 1993, Morgan *et al*., 1998, de Roo *et al*., 1998, Torri *et al*., 1999). Onstad (1984) and Govers *et al*. (2000) have developed empirical equations to estimate surface storage resulting from random surface roughness and they have recognized that this effect has larger significance immediately after cultivation. Large random roughness may have such a strong effect, that it can temporarily decouple the relationship between slope angle and runoff (NEARING *et al.*, 1997). Erosion models usually consider random roughness as a deterministic property of the soil surface. Only a few model approaches have been reported where the spatial organisation of rill and interrill runoff networks were represented stochastically (KIRKBY *et al.*, 1993. FAVIS-MORTLOCK, 1998, DABROUX *et al.*, 2001). Our results also confirm the excessive importance of surface roughness on erosion and the relative importance of rills in shortcutting flow paths.

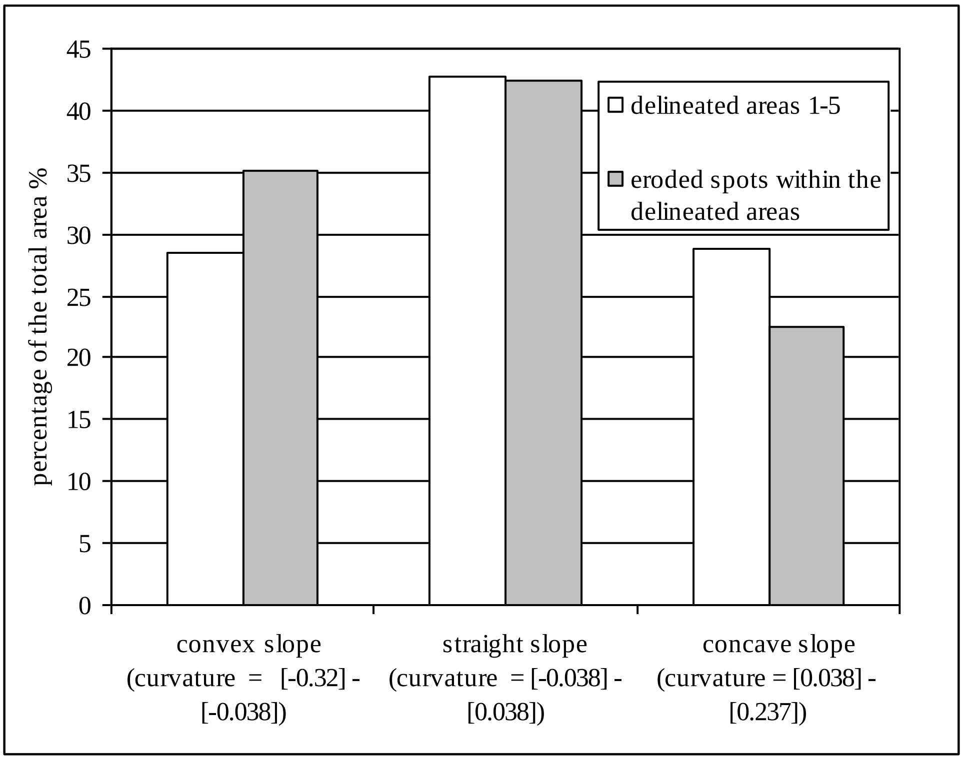


*Figure 4*. Distribution of slope shape categories within the delineated areas and within the strongly eroded spots (eroded spots are 15.6 % of the total area within the delineated erosion fields 1-5.)

*Table 5*. Decrease of flow lengths and increase of slope angles due to rill formation

| Erosion area No. | Longest flow length with random roughness only (m) | Decrease of flow lengths due to the superimposed rills (m) | Decrease of flow lengths (%) | Maximum height differences within the areas (m) | Mean slope without rills (%) | Mean slope with rills (%) |
|---|---|---|---|---|---|---|
| 1 | 1462.3 | 112.4 | 7.7 | 89.18 | 6.10 | 6.61 |
| 3 | 1482.4 | 230.4 | 15.5 | 87.06 | 5.87 | 6.95 |
| 2 | 985.8 | 106.6 | 10.8 | 61.06 | 6.19 | 6.95 |
| 4 | 1282.3 | 153.9 | 12.0 | 56.92 | 4.44 | 5.04 |
| 5 | 1375.4 | 99.0 | 7.2 | 88.49 | 6.43 | 6.93 |
| Average | 1317.6 | 140.5 | 10.6 | 76.5 | 5.81 | 6.50 |

**Conclusions**

The soil and phosphorus load from the Somogybabod study catchment in June 2006 was mainly due to rill erosion. Since aridification and frequency of flash floods increase in the southern sub-watershed, phosphorus load from there to Lake Balaton may increase in the future. Extremely large phosphorus load can be expected from the slope shoulders and backslopes in summer months during heavy thunderstorms and due to erodible soils derived from sandy loess. Rill erosion plays significant role in these excessive events. Approximately 90 % of the load could be attributed to bulk erosion from rills in the investigated period. Rills connect the most erodible spots of arable land directly to the stream network thus they switch the catchments into a higher P load mode than they were before without rills. These spots are often eroded down to the calcareous parent material which leads to the conclusion that the erosion from these spots have been around 60–90 $t.ha^{-1}.yr^{-1}$ on the average for the last two hundred years. It implies too that riling is a permanent phenomenon in this environment. Rills short cut the dispersed flow of surface water caused by the random roughness thus, they significantly increase real slopes hence erosion and phosphorus load. Our results very much support the idea of VANDEKERCKHOVE *et al*. (2004) who suggested catchment scale monitoring of rills as indicator and potential policy tool in soil protection efforts.

Random roughness is an important model component which is suitable to incorporate many effects into erosion modelling (surface cover, crop development, soil settling after cultivation, real random roughness of the surface, digital elevation model uncertainty etc.). It may enable to simplify erosion models by reducing the number of core variables that drive the process.

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# THE INFLUENCE OF HYDROLOGICAL CONNECTIVITY ON NUTRIENT DYNAMICS OF A DANUBE OXBOW (DANUBE-DRÁVA NATIONAL PARK, HUNGARY)

*Edit ÁGOSTON-SZABÓ – Mária DINKA – Károly SCHÖLL*

Danube Research Institute of the Hungarian Academy of Sciences, H-2163 Vácrátót, Hungary, e-mail: edit@mta-dki.hu

**Abstract:** Nutrient and carbon concentrations of the Külső-Béda, which is an oxbow on the right bank of the Danube in the active floodplain area of Béda-Karapancsa nature protection area (Danube-Dráva National Park, Hungary), were investigated and compared with the main arm of the Danube at different water levels. The concentration of the nitrate, phosphate, total dissolved nitrogen, total dissolved phosphorus was lower, while the concentration of the dissolved organic carbon and dissolved total carbon was higher in the oxbow than in the main arm of the Danube. The lower nutrient and the higher carbon concentrations of the oxbow as compared to the main arm suggested that the oxbow was a sink for nutrients and a source of carbon and underlined its role fulfilled in nutrient reduction.

**Key words:** nitrogen, phosphorus, carbon, oxbow, Danube

## Introduction

Natural river-floodplain ecosystems are characterised by a mosaic of aquatic habitats with different hydrological connectivity with the main arm and one of the most important ecological functions of these ecosystems is the exchange of nutrients between the water and its adjacent floodplain (JUNK *et al.,* 1989, MALMQVIST & RUNDLE 2002).

Oxbows are floodplain water bodies isolated for most of the year from the main arm, characterised by high nutrient retention potential, which is mainly realized by the denitrification, sedimentation processes or uptake of nutrients by the aquatic organisms (GLIŃSKA-LEWCZUK 2009), their nutrient purification capacity can be negatively affected by the anthropogenic interventions such as the river regulation (HEIN et al. 2005). The human activity-linked nutrient excess with agricultural, industrial, domestic or other origins results in an increasing algal biomass and eutrophication of the waters, which can lead to the water quality problems (MURDOCK & DODDS 2007). Oxbows serve as buffers between the agricultural uplands and rivers (especially for the $NO_3^-N$) and provide important ecological services (MITSCH *et al.,* 2001, GLIŃSKA-LEWCZUK 2005, SCHRAMM *et al.,* 2009). Their water chemistry is highly influenced by the water level fluctuation of the main arm of the river and by the movement of the alluvial groundwater (GLIŃSKA-LEWCZUK 2005).

**The aim of this work** was to study the role of Külső-Béda oxbow in the nutrient retention by examining the spatial and temporal changes in the nutrient concentration of the water at different water levels of the main arm of the Danube.

**Material and Methods**

The Külső-Béda (BDU) is an oxbow with high natural value, situated on the right bank of the Danube, in the Béda-Karapancsa Landscape Protection Area of the Danube-Dráva National Park. It has a limited connectivity with the main arm, with junctions with the Danube at rkm (river-kilometre) 1440.5 (upstream) and the mouth at rkm 1437.5 (downstream). The threshold level of the upstream is 6.3 m (at gauge of Mohács, rkm 1447) and that of the downstream is only 1.2 m (at rkm 1447). Its open water area is 4 km long, 90 m wide on average and about 2.5 m deep (Fig. 1).

The investigations were carried out at six sampling sites: BDU1, BDU2, BDU22, BDU3, BDU4, BDU5, situated at 450-750 m distance from each other (Fig. 1) and on five sampling dates: 25 June, 29 August, 25 October in 2007 and 18 March, 10 June in 2008 (Fig. 2).

Reference samples were taken simultaneously from the Danube at rkm 1447. The temperature, pH, electrical conductivity and oxygen concentration of the water were determined *in situ* with Multi 340i meter (WTW). The total organic carbon (TOC), dissolved organic carbon (DOC), total dissolved nitrogen (TDN) concentrations were determined in the laboratory by TOC analyser (Elementar-liqui-TOC). The phosphate ($PO_4^{3-}$-P), nitrate ($NO_3^-$-N), the total dissolved phosphorus (TDP) concentrations of the water were determined in the laboratory by standard analytical methods (GOLTERMAN *et al*., 1978).

Data was analysed statistically using Microsoft Excel and Statistica program packages.

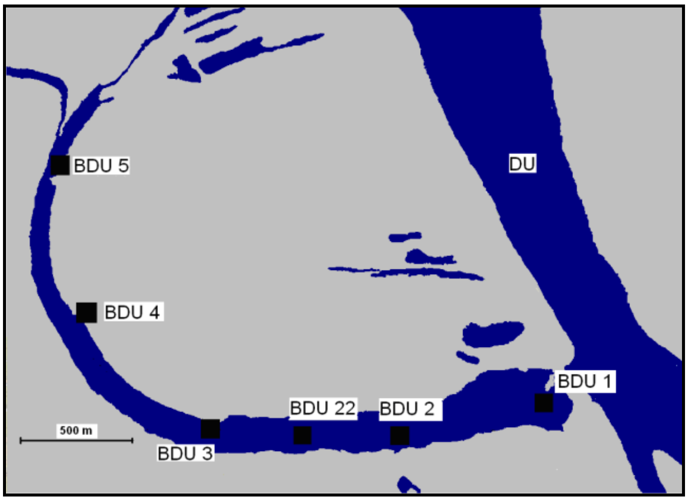


*Figure 1.* Sampling sites at Külső-Béda oxbow: BDU (Danube: DU)

**Results and discussion**

The changes in water level of the Danube (DU) (rkm 1447) and the sampling dates are presented in Fig. 2. The water level of the DU ranged from 1.36 m to 7.76 m and exceeded the downstream threshold of the BDU: 1.20 m at rkm 1447) during the investigation period (25.07.2007-10.06.2008).

The upstream threshold level of the BDU: 6.30 m at rkm 1447 was exceeded by the water level of DU only during a 9 day period, between 11.09.2007-19.09.2007, when the highest magnitude of flood pulse occurred. The highest flow peak was observed on 15.09.2007 (Fig. 2).

The chemistry of floodplain lakes are mainly determined by the interplay between the chemical parameters and the hydrology of the main arm (VAN DEN BRINK *et al.,* 1993). According to our results the hydrologic conditions, which are important controlling factors of the nutrient retention capacity of riverine wetlands, exerted an important influence on the chemical parameters of the BDU.

The temperature of the oxbow varied between 9.3±0.7 and 25.9±0.5 C° (n=6) during the investigation period (Fig. 3).

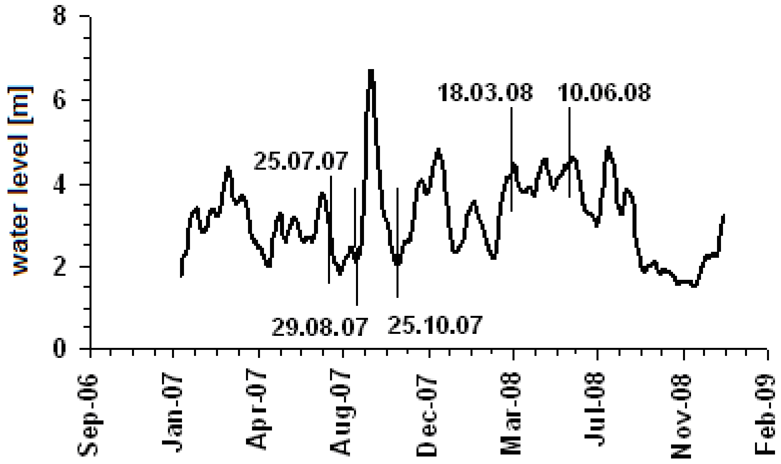


*Figure 2*. The hydrograph for the Danube (at rkm 1447) and the sampling dates

At low water levels, the electrical conductivity of the oxbow water was higher, while at high water levels and after the highest flood event it was lower than that of the main arm. The highest conductivities were measured after the flood pulse, which suggest that the flood event remarkably influenced the chemistry and the habitat conditions in BDU (Fig. 3).

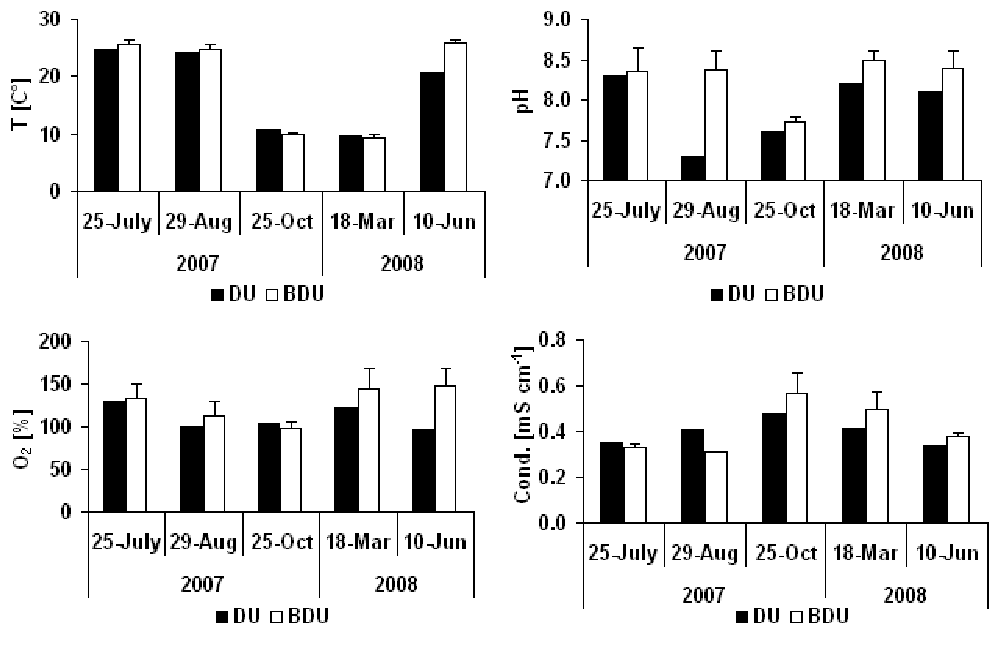


*Figure 3*. The temperature, pH, oxygen concentration and electrical conductivity of the Külső-Béda (BDU) (mean ±SE, n=6) and of the main arm of the Danube (DU) at rkm 1447

The pH of the oxbow water was higher than in the main arm. The lowest pH values in the oxbow were measured after the highest flood pulse (Fig. 3).

Higher oxygen concentrations were measured in the oxbow water than in the main arm (except 25 Oct. 2007) (Fig. 3). The higher $O_2$, conductivity, pH in BDU than in DU probably ensure better productive conditions in BDU.

The $NO_3^-$-N concentration varied between 0.03±0.10 and 1.20±0.50 mg $l^{-1}$ (n=6) in the BDU, and it was remarkably lower than that of the DU (Fig. 4). The highest $NO_3^-$-N concentration was recorded in spring both in BDU and DU, when the TDN fraction was dominated by the $NO_3^-$-N.

The TDN concentration was also lower in the BDU (1.02±0.20 - 1.62±0.10 mg $l^{-1}$, n=6) than in the DU (Fig. 4).

In a similar way as the concentration of the examined N forms, the $PO_4^{3-}$-P and TDP concentrations of DU exceeded that of the BDU (Fig. 4).

According to VAN DIJK *et al.* (1994) a concentration of less than 0.15 mg $l^{-1}$ P and 2.2 mg $l^{-1}$ N is necessary for preventing the eutrophication.

One of the main driving forces for the nutrient retention of floodplain waters is the hydrological connectivity (GLIŃSKA-LEWCZUK 2005). The high $NO_3^-$-N, $PO_4^{3-}$-P , TDP, TDN of the oxbow water in spring of 2008 (Fig. 4), associated with high water level (Fig. 2) and a higher degree of connectivity could be the result of the nutrient input from the river to the oxbow.

During our investigation period lower $PO_4^{3-}$-P, TDP concentrations were measured after the highest flood pulse as compared to the other sampling date (Fig. 4), which might be explained by the delivery of P to the adjacent floodplain during the flood event (TOCKNER *et al.,* 1999).

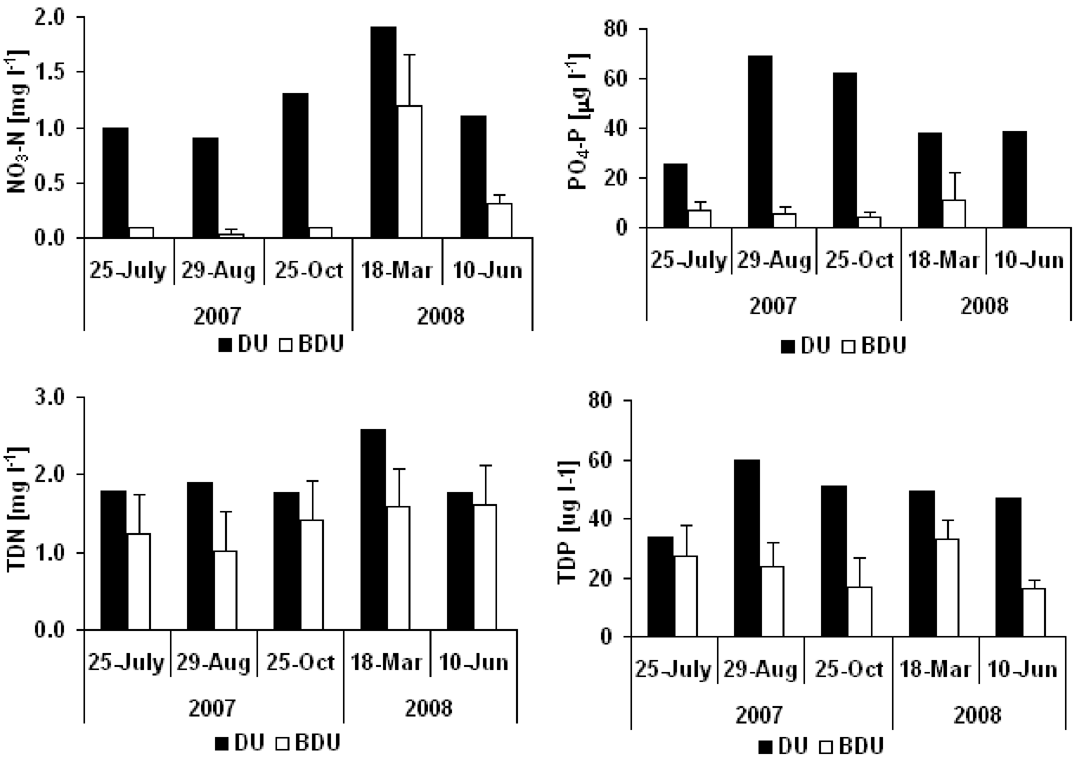


*Figure 4*. The concentrations of the nitrogen and phosphorus forms in the Külső-Béda (BDU) (mean ±SE, n=6) and in the main arm of the Danube (DU) at rkm 1447

The TOC (6.7±1.0–31.0±12.3, n=6) and DOC (5.2±0.3–16.8±1.8%, n=6) concentrations in BDU exceeded those of the main arm (Fig. 5). The highest DOC was recorded after the flood pulse, which might be the result of the carbon import from the adjacent floodplain area. The relationship between the amount of DOC released from inundated floodplain surface and the magnitude of flood has been described by THOMS (2003) and, according to his results, the floodplain surface significantly influences the potential supply of DOC.

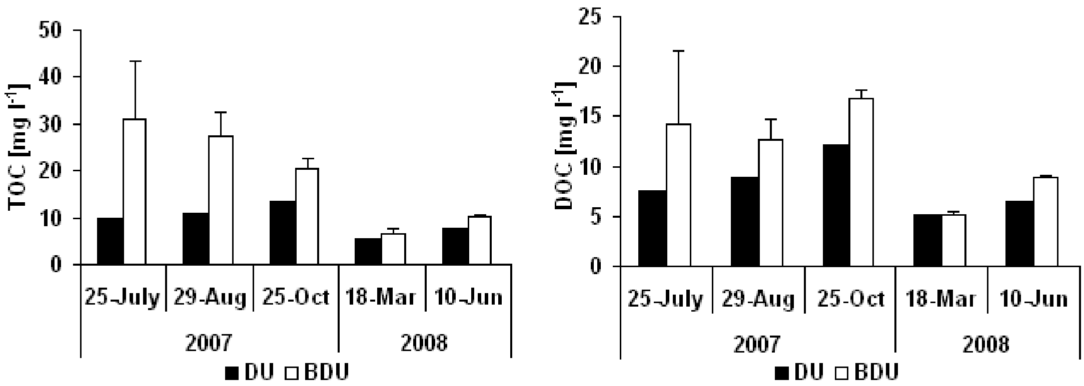


*Figure 5*. The concentration of the carbon forms of Külső-Béda (BDU) (mean ±SE, n=6) and of the main arm of the Danube (DU) at rkm 1447

The allochthonous DOC (released from the sediment or leaf litter lying on the floodplain during the flood event) represent an important energy source for the aquatic organisms (CHAUVET, 1997). Once they enter the water, they are immediately assimilated by micro-organisms which use up oxygen at a faster rate than it can be replenished (HOWITT *et al.,* 2007). This could be the reason for the lower dissolved oxygen concentrations measured after the flood pulse as compared to other sampling date.

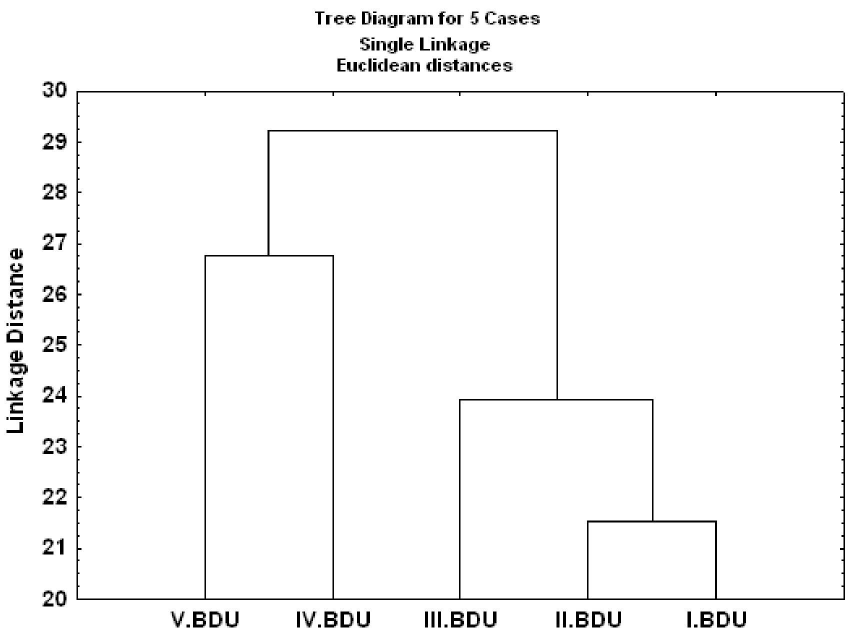


*Figure 6*. The cluster analyses of the Külső-Béda (BDU) sites at different sampling times (I: 25-July-2007, II: 29-Aug-2007, III: 25-Oct-2007, IV: 18-Mar-2008, V. 10-June-2008)

The results of cluster analyses reflected the separate grouping of the BDU sites at high water level (IV.BDU: 18-Mar-2008 and V.BDU: 10-June-2008) and at low water level (I.BDU: 25-July-2007 and II.BDU: 29-Aug-2007) and after the highest flood event, respectively (III.BDU: 25-Oct-2007) (Fig. 6). The separation of the BDU sites after the flood pulse from the rest of the sampling date suggested the notable influence of this flood event on the water chemistry and habitat conditions. The effect of water levels on the habitat conditions of the Külső-Béda oxbow was also reported by ÁGOSTON-SZABÓ *et al.* (2010).

**Conclusions**

The lower N and P and the higher C concentrations of the oxbow as comparing to the main arm, suggested that the oxbow was a sink for nutrients and a source of carbon and confirmed its role in nutrient retention and in the energy transport across the floodplain-river system.

The changes in water level and the magnitude of flooding remarkably influenced the exchange processes between the river-oxbow and adjacent floodplain area and consequently the nutrient and carbon cycle in river-floodplain ecosystem. The nutrient retention capacity of the oxbow is highly dependent on the degree of connectivity and on the water level fluctuation of the main arm.

Our results can provide scientific bases for floodplain managers in their work of floodplain restoration and water quality protection.

**Acknowledgments**

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# WATER QUALITY IN JIU RIVER BASIN

*Ana Maria DODOCIOI*[1] *– Romulus MOCANU – Marian DOBRE – Alexandru TUDOSIE*

[1] University of Craiova, Faculty of Agriculture, Dept. of Plant nutrition, Libertatii street, Craiova 200585, Romania; e-mail: ana_m3310@yahoo.com

**Abstract**

The Jiu River basin is located in South – West part of Romania, at $43^0$, 45` - $45^0$, 30` northern latitude, $22^0$, 34` - $24^0$, 10` eastern longitude. The altitude is 1,649 m in the northern part and 24.1 m in the southern part where it flows into the Danube. The water has large industrial and municipal wastewater loads at the lower reaches. Chemical water quality indicators of Jiu River were measured at three locations and on five dates during the year 2010 and their trends were the following:

- dissolved oxygen exceeded 9 mg/litre in the upper part and it decreased downwards, at the last location being under optimum (8.5-6.5 mg/litre) and it decreased between first the measurement in January and the last one in November 2010;
- The ammonium nitrogen was under the critical limit in all samples;
- The nitric nitrogen exceeded tolerance level (1 mg/litre) at all points and times, being between 1.17 – 2.35 mg/litre;
- The nitrous nitrogen exceeded the tolerance level (0.01 mg/litre) in all samples, reaching 0.016-0.036 mg/litre;
- Sometimes Ni and Pb contents exceeded tolerance levels due to the fact that the Jiu River crosses an area with open cut and underground mines and due to accidental industrial pollution.

**Keywords:** Jiu River basin, chemical water quality indicators, tolerance levels, dissolved oxygen, ammonium, nitrates, nitrites, heavy metals.

**Introduction**

European waters are under an increasing pressure with continuous demand for good quality fresh water in sufficient quantity for all needs.

In order to improve the aquatic environment and water quality, the European Community set out the Directive 2000/60/CE of the European Parliament and of the Council of 23.10.2010 also called the Water Framework Directive (WFD).

The objective of this directive was to establish a framework for the protection of inland waters, transitional waters, coastal waters and groundwater by ensuring an increasing protection and the improvement of the aquatic environment, especially by the progressive reduction in emissions of toxic substances.

WFD is considered a progressive and innovative legislative measure to improve the surface waters across Europe. It requires ecological evaluations of a wide series of indicators of environmental pressures and ecological stresses related to the degree of deviation from the reference values (COST 869 Final Report).

For a comprehensive approach, it is necessary to perform an analysis of the characteristics of a river basin and of the impact of human activities on it, as well as an economic analysis of the water use. The directive also states the need for preventing or gradually eliminating the release of dangerous substances from agriculture and industry into surface waters (CUTTLE, 2006; NEWELL, 2010). This is why we considered it necessary to analyse the way the stipulations of WFD are respected within the Jiu River basin.

**Materials and methods**

The Jiu River basin is located in South – West Romania, at $43^0$, 45\` - $45^0$, 30\` northern latitude, $22^0$, 34\` - $24^0$, 10\` eastern longitude. The altitude is 1,649 m in the North and 24.1 m in the South where it flows into the Danube. It has a drainage area of 10,800 $km^2$ (4.2% of the country) of which 4,935.16 $km^2$ is arable. The watershed has an approximate length of 260 km, the width being 60 km in the upper part and 20 km in the lower part, crossing 4 districts out of 40: Hunedoara, Gorj, Mehedinti and Dolj. The population within this basin is 1,461,661 (6.6% of Romanian population). The Jiu River has 54 tributaries.

The rocks of this basin consist of silica and lime, the age of the deposits being Miocene, Pliocene and Quaternary.

The water resources are 4,059 million $m^3$/year of which 2,109.5 $m^3$ are used. Within Jiu River basin there are 67 lakes with a useful volume of 147.61 million $m^3$/year and 69 paddles. The main soil types are: chernozems, clay illuviated soils, waterlogged soils, alluvial soils and sandy soils (BOENGIU, 2007).

The applied fertilizer quantities within this area in 2010 were: 7.9 kg.N/ha, 2.11 kg.P/ha and 743 kg manure/ha.

The number of domestic animals number was 458,800, with an average of 0.4 animal/ha (DODOCIOIU, 2011).

Seventy eight percent of the total number of villages and communes in the area are vulnerable to nitrate pollution because of random disposal of animal manure and the lack of treatment facilities and special channels for eliminating animal waste (MOCANU, 2011).

When we consider the complexity of this river basin, taking into account its geography, agriculture, mining industry in the upper mountains and hills, the chemical industry and coal-fired power stations of Craiova city , there are high levels of pollutants which reach the Jiu River with or without treatment. Paroseni, Rovinari, and Turceni power plants in the northern part of the river basin burn lignite. Due to the high water volume they use for cooling, the river temperature increases by at least $7^0$C above normal.

Within the Podari and Malu Mare zone, water is released from the Isalnita Power plant (10 km from Craiova city), Doljchim S.A. Craiova (a chemical Plant which produces nitrogen and complex fertilizers) and residual water from Food Industry Platform in Podari. In 2008 the Isalnita Power plant released into Jiu River the following quantities of pollutants: 1,619.83 t suspensions, 12,396.82 t dry residues, 926.78 t chlorures, 11.26 t ammonium, 249.47 nitrates. In 2008 Doljchim SA released in its waste water the following quantities of pollutants into Jiu River: 173.03 t $BOD_5$, 241.37 t suspensions, 3,333.4 t dry residues, 18.81 t ammonium, 130.09 t nitrates. Oltenia Water Company released into Jiu River in 2008, along with municipal wastewater of Craiova and Filiasi, without any treatment the following pollutants: 2,813.95 t $BOD_5$, 6,377.71 t suspensions, 30,043.87 t dry residues, 31,211.2 t chlorures, 2,183.29 t ammonium and 341.97 t nitrates (County Environmental Agency data). This is the reason why we considered it necessary to quantify the main quality indicators of the Jiu River. For this purpose we investigated the water quality in 2010 at 3 points of the river basin as follows:

- Racari – the zone just after the mountain section (after coal mines);
- Podari – 12 km downward of Craiova (the main city) after main industry facilities;
- Malu Mare – 20 km downward of Craiova, after Craiova and Podari Food Industry facilities.

From these three locations water samples were taken using SR ISO 5667-6/2000 standard method at 6 dates during the year: 14.01; 09.03; 03.05; 06.07; 09.09; 10.10.2010. The water samples were analyzed as follows:

- pH - SR ISO 10523/2009 standard;
- Dissolved oxygen - SR EM 25813/2000 standard;
- Ammonium, N-$NH_4$ - STAS 8683/1970 standard;
- Nitrates N-$NO_3$ - SR ISO 7890-1/1991 standard;
- Total nitrogen - SR ISO 10048/2001 standard;
- Total Phosphorus - SR EM 6878/2005 standard;
- Calcium and Magnesium - STAS 3662/1990 standard;
- Heavy metals (dissolved Cr, As, Cd, Ni, Pb) - SR EM ISO 17294-2/2005 standard.

**Results and discussion**

The pH data analysis show normal values between 6.55-7.75 for all determination points and dates. Usually, the pH value has not changed much between March and October. Regarding pH values, the water quality is first class.

The dissolved oxygen content had close values in January and March and decreased afterwards. This is due to the intensification of the microbiological processes and phytoplankton and zooplankton activity in the river. On 14 of January and 9 of March, the dissolved oxygen content was better then the threshold 9 mg/litre, being of 11.60-12.08 mg/litre. From this point of view the Jiu River water was classified within first and second quality classes.

The ammonium content was low in all samples at Racari (0.10-0.39 mg/litre), being under tolerance level due to the small concentrated emissions in the mountain and hilly zone (in Northern part of the basin). The water belongs to the first (this is the best) quality there.

At Podari and Malu Mare, the ammonium content in Jiu River was higher (of 0.13-0.85 mg/litre) because of loading from Doljchim S.A., Isalnita Power plant and Podari Food Industry.

At Podari, the ammonium concentration exceeded tolerance level at the first sampling date, being 0.43 mg/litre, but it remained under tolerance level later; at this point the water was classified as first class.

At Malu Mare, the ammonium content was high yet under tolerance level. The concentration (0.85 mg/litre) exceeded the tolerance level on 06.07.2010 only. Regarding the ammonium content, the water of Jiu River is classified as first and second class.

*Table 1*. Water quality of Jiu River based on dissolved oxygen, pH and nutrients

| Point | Dates | Indicators/ min or max. tolerance levels and units | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | Dissolved $O_2$ | pH | N-$NH_4$ | N-$NO_3$ | N-$NO_2$ | Total N | Total P |
| | | 9 mg/L | 6.5-8.5 | 0.4 mg/L | 1 mg/L | 0.01 mg/L | 1.5 mg/L | 0.015 mg/L |
| Racari | 14.01.2010 | 11.06 | 7.6 | 0.13 | 0.17 | 0.02 | ‹3 | 0.04 |
| | 09.03 | 11.76 | 7.51 | 0.12 | 1.18 | 1.01 | ‹3 | 0.08 |
| | 03.05. | 9.03 | 7.61 | 0.19 | 1.48 | 0.01 | 1.86 | 0.04 |
| | 06.07. | 8.09 | 7.40 | 0.39 | 1.12 | 0.01 | 1.84 | 0.04 |
| | 09.09 | 7.61 | 7.62 | 0.18 | 0.91 | 0.02 | 1.51 | 0.07 |
| | 10.10. | 9.70 | 7.48 | 0.10 | 1.01 | 0.02 | 1.49 | 0.05 |
| Podari | 14.01.2010 | 11.28 | 7.75 | 0.43 | 1.12 | 0.03 | 3.7 | 0.03 |
| | 09.03 | 12.08 | 7.69 | 0.23 | 1.29 | 0.02 | ‹3 | 0.05 |
| | 03.05. | 9.81 | 7.75 | 0.38 | 1.33 | 0.02 | 2.29 | 0.05 |
| | 06.07. | 8.78 | 7.16 | 0.34 | 1.73 | 0.02 | 1.85 | 0.04 |
| | 09.09 | 7.88 | 7.53 | 0.32 | 1.65 | 0.02 | 2.23 | 0.06 |
| | 10.10. | 9.44 | 7.35 | 0.131 | 1.21 | 0.02 | 1,62 | 0.05 |
| Malu Mare | 14.01.2010 | 8.91 | 7.46 | 0.29 | 3.07 | 0.01 | 4.1 | 0.24 |
| | 09.03 | 10.18 | 7.42 | 0.26 | 2.17 | 0.02 | ‹3 | 0.09 |
| | 03.05. | 7.16 | 7.28 | 0.34 | 2.11 | 0.02 | 2.97 | 0.09 |
| | 06.07. | 6.96 | 6.55 | 0.85 | 2.35 | 0.04 | 3.81 | 0.11 |
| | 09.09 | 7.73 | 7.25 | 0.27 | 2.51 | 0.03 | 3.39 | 0.06 |
| | 10.10. | 10.10 | 7.53 | 0.26 | 1.78 | 0.03 | 2.5 | 0.09 |

*Table 2*. Water quality of Jiu River based on metal ion contents

| Point | Dates | Indicators/ max. tolerance levels and units | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | $Ca^{2+}$ | $Mg^{2+}$ | Cr | As | Cd | Ni | Pb |
| | | 50 mg/L | 12 mg/L | 2.5 µg/L | 7.2 µg/L | 1 µg/L | 2.1 µg/L | 1.7 µg/L |
| Racari | 14.01.2010 | 40.20 | 8.06 | 1.20 | <2.0 | 0.20 | 1.90 | 0.42 |
| | 09.03 | 37.70 | 7.54 | 1.50 | <2.0 | 0.20 | 1.50 | 0.40 |
| | 03.05. | 36.90 | 7.40 | 1.50 | <2.0 | 0.20 | 1.70 | 0.31 |
| | 06.07. | 38.60 | 7.66 | 1.60 | <2.0 | 0.20 | 1.40 | 0.40 |
| | 09.09 | 48.35 | 9.57 | 0.74 | <2.0 | 0.20 | 0.63 | 0.30 |
| | 10.10. | 28.00 | 3.3 | 1.80 | <2.0 | <0.20 | 2.30 | 1.5 |
| Podari | 14.01.2010 | 43.09 | 8.62 | 1.40 | <2.0 | <0.20 | 3.40 | 0.69 |
| | 09.03 | 55.43 | 11.10 | 1.10 | <2.0 | <0.20 | 1.50 | <0.30 |
| | 03.05. | 46.33 | 9.37 | 1.30 | <2.0 | <0.20 | <1.50 | <0.30 |
| | 06.07. | 43.79 | 15.16 | 1.30 | <2.0 | <0.20 | <1.50 | <0.30 |
| | 09.09 | 28.96 | 5.79 | 1.20 | <2.0 | <0.20 | <1.50 | <0.30 |
| | 10.10. | 37.50 | 7.5 | 1.30 | <2.0 | <0.20 | 3.0 | 0.73 |
| Malu Mare | 14.01.2010 | 59.78 | 11.16 | 1.10 | <2.0 | <0.20 | 2.6 | 2.6 |
| | 09.03 | 45.94 | 9.19 | 1.30 | <2.0 | <0.20 | 2.1 | 1.0 |
| | 03.05. | 79.58 | 15.92 | 1.00 | <2.0 | <0.20 | 1.7 | 0.87 |
| | 06.07. | 48.65 | 9.73 | 1.00 | <2.0 | <0.20 | 1.1 | 0.30 |
| | 09.09 | 43.82 | 7.86 | 1.40 | <2.0 | <0.20 | 3.3 | 0.57 |
| | 10.10. | 43.60 | 9.56 | 1.60 | <2.0 | <0.20 | 1.2 | 0.43 |

The nitrate content exceeded tolerance level in most of the samples. The lowest values of 0.17-1.48 mg N/litre were recorded at Racari due to the small water volume of tributaries and their low nitrate content. The highest nitrate content of 1.78-3.07 mgN/litre was recorded at Malu Mare because of the influence of waste from large factories. With regard to the nitrate content of the water, Jiu River was classified as second and third class.

The nitrite content also exceeded tolerance level (0.01 mg/litre) in all points and all determination dates with values between 0.02-1.01 at Racari, 0.02-0.03 at Podari and 0.01-0.04 mgN/litre at Malu Mare. From this point of view, the Jiu River was considered as second and third class.

The total nitrogen content was high, too, at all points. It exceeded tolerance level (1.5 mg N/litre) at all dates of determination. The highest values were recorded, also, at Malu Mare, of 2.5-4.1 mgN/litre. As a result, the waters were classified as second and third quality class.

The total phosphorus content exceeded tolerance level in all points and at all dates of determination. Also, higher values were recorded at Malu Mare (0.06-0.29 mgP/litre). The water was considered as second and third class regarding this indicator.

The content of base cations, $Ca^{2+}$ and $Mg^{2+}$ have normal values at Racari and Podari, being under tolerance levels and the water is first class. At Malu Mare, the $Ca^{2+}$ content exceeded tolerance level only at two dates, 14.01 and 13.05. and the water is second class.

The heavy metal content is, usually, under tolerance levels. Specifically, the contents of Cr, As and Cd are under tolerance level for all points and dates and the water is first class.

The Ni content recorded high values at 10.10.2010 at Racari, at 14.10 – Podari and at 14.01 – Malu Mare, probably because of uncontrolled loads. The lead content is under tolerance level for all dates and points excepting Malu Mare at 14.01 when 2.6 µg/L was recorded over tolerance level (1.7 µg/L) because of uncontrolled loads from a nearby tannery.

**Conclusions**

Analysis of water samples from Jiu River basin was performed at three locations: Racari, Podari and Malu Mare at 6 dates. The following conclusions can be drawn:

- the pH of the water in Jiu River has normal values of 6.55-7.75 which do not exceed tolerance levels (6.5-8.5);
- the dissolved oxygen content has high values at the beginning of the investigation period (14.01) and then decreases;
- the ammonium and nitrate contents were under tolerance levels at Racari, but exceeded acceptable levels at Podari and Malu Mare due to Doljchim chemical plant, Isalnita power plant and waste waters from Craiova city;
- the total nitrogen and total phosphorus exceeded tolerance levels in all locations and at all dates of investigation;
- the heavy metal contents were, in general, between normal limits. Thus, Cr, As and Cd had acceptable values in all points and dates. The Ni content exceeded tolerance level at first sampling at Podari and Malu Mare; the other values were under the critical level;
- on the basis of these analyses it can be seen that the water of the Jiu River is not of the highest quality (second class)

On the basis of actual recommendations, a modern wastewater treatment plant for Craiova city is under construction and the largest pollutant, the Doljchim chemical plant is upgrading its technology.

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# GROUNDWATER QUALITY MONITORING IN AN EXPERIMENTAL FIELD STATION IN SOUTH BULGARIA

*Dimitranka* STOICHEVA[1] – *Milena* KERCHEVA[1] – *Venelina* KOLEVA[1] – *Tsetska* SIMEONOVA[1]

[1]N. Poushkarov Institute of Soil Science, 7 Shosse Bankya Str., 1080 Sofia, Bulgaria, e-mail: dstoicheva@abv.bg

**Abstract**

The aim of this study is to present results on ground water quality monitoring realized on the territory of a small watershed at the experimental station Tsalapitsa in South Bulgaria, under Fluvisol. Fluctuation of the shallow groundwater table was monitored at three permanently built piped drilling wells and samples for chemical analysis were taken monthly. The assessment of groundwater chemical composition is based on data for the period 2005 - 2010 years, when different types of agricultural crops were grown and also there was a change in the loading with fertilizers of the watershed. The results were compared with previous 12-year monitoring period 1992-2004.

Nitrate content in the groundwater is influenced by the reduced anthropogenic loading with fertilizers and there can be seen a decreasing trend in nitrate concentration during the last years from the monitoring period when nitrate concentrations are around and below MPCL. The highest nitrate concentrations were measured during spring-summer months. A correspondence was observed between the dynamics of $NO_3^-$ and $K^+$ content and the same trend of temporal fluctuation of both elements. This suggests that the variation in these elements under this soil type have one and the same origin. Calcium concentrations in groundwater vary in considerably short range, although the variation is not so significant. Fluctuation corresponds to that of the nitrate content.

**Key Words:** chemical composition, groundwater, agriculture, nitrate pollution, Fluvisol, South Bulgaria.

## Introduction

Groundwater is one of the most valuable water resources for both municipal supplies and for rural settlements, mainly in river basins, where many Bulgarian cities and villages are located. Most of these territories are used for intensive agriculture. The main soil type in these areas is Alluvial-Meadow soil (Fluvisols). These soils cover 8,5 % of the country. Fluvisols are young soils and their formation process is influenced by the shallow groundwater with a direct hydraulic connection with the rivers. The chemical composition of groundwater depends on relatively constant factors (soil and geology), dynamic natural factors (climate) and anthropogenic impact (fertilizers, septic systems and livestock wastes). Overloading with fertilizers is the most probable source of excessive nitrate levels in the groundwater. Several groundwater monitoring programs have been initiated for the last few years to determine the best agricultural management practices and to protect groundwater from diffuse pollution from agricultural sources thus, to obey the requirements of the Nitrate Directive (91/676/EEC) and Council Directive 98/83/EC.

Although some chemical quality standards have been established at European level for particular issues - nitrates, pesticides and biocides, this could not ensure the society's demand for pure water. The requirement for groundwater monitoring is set in order to detect any negative changes in chemical composition. This should ensure the protection of groundwater from all contamination, according to the principle of minimum anthropogenic impact (Water Framework Directive, 2000).

The purpose of this study is to present results on monitoring the dynamics of some chemical elements in the groundwater in a field experimental station on Fluvisol and to evaluate the influence of changing anthropogenic loading on the groundwater chemical composition.

## Material and Methods

This study was conducted in a small watershed at the experimental station of the village of Tsalapitsa in South Bulgaria on Alluvial-Meadow soil (Fluvisol). The experimental station of Tsalapitsa (Lat= N 42°11', Long=E 24°32.5') is situated in the west part of the Thracian Plain, near the town of Plovdiv. Site selection was based on some soil characteristics, which render the soil as highly susceptible to nitrate leaching into shallow groundwater, as well as the availability of long-term experimental data. This allows adequate assessment of the changes in the groundwater quality due to the anthropogenic impact. Long-term datasets are very well suited for supplementary and comparative purposes.

Hydrogeological research on the territory shows high spatial heterogeneity of the thickness and distribution of the alluvial materials (Stoichev *et al.,* 1980; Mateva *et al.,* 1982). Dominant materials are mainly Pliocene and Quaternary sediments. Under the root zone at a depth of around 150-200 cm a layer of grey coarse-grained sand and silt is observed. This causes a lack or very slow capillary rise of water and soluble chemicals from shallow groundwater to the active root zone. The depth of groundwater level varies between 2.6 and 4.8 m (during the period of research varied from 2.8 to 5.5 m), The first aquifer consists of light-brownish slightly compact sandy clay evidenced by the hydraulic slope (j=0,0016) and conductivity of that soil layer ($K_{sat} = 7.23 \cdot 10^{-5}$ m/s). The surface soil could be characterized with low water holding capacity and rapid water exchange between the soil layers, which is a precondition for migration of the dissolved materials through the soil profile.

Fluctuation of the shallow groundwater table was monitored at three (№5, №6, №7) permanent wells (∅ 220 mm) and samples for chemical analysis were taken monthly. The assessment of groundwater chemical composition was based on data for the recent period of 2005-2010, when different agricultural crops were grown and also there was a change in fertilizer use in the small watershed. Comparisons were made with the period of 1995-2004.

The region is situated in a moderately continental climatic sub-zone influenced by Mediterranean climate. The average annual precipitation for 1961-1990 was 480 mm with two maxima – in May and November (Figure 1). Total temperature amounts to about 4000 °C for temperatures above 10 °C. Mean annual air temperature for this region is 12.2 °C. The comparison between monthly levels of precipitation during the study period (2005-2010) and those for 1961-1990 (reference climate) and 1995-2004, show a significant increase of the precipitation in summer and autumn months of the recent years (Figure 1). This is due to heavy rains causing several floods in the region.

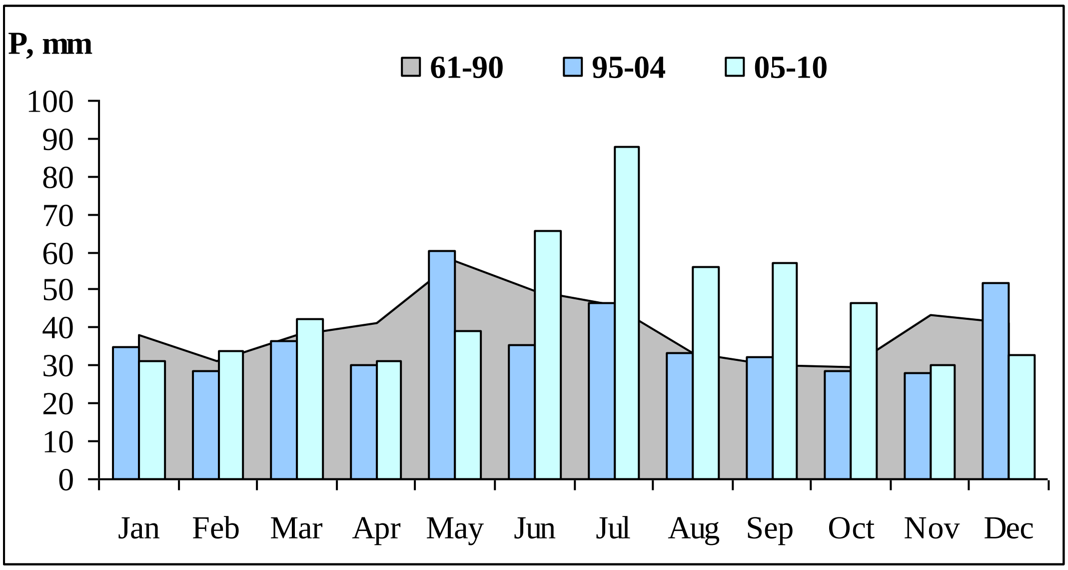



*Figure 1*. Monthly precipitation sums for the periods 2005-2010 (05-10), 1995-2004 (95-04), 1961-1990 (61-90) in Tsalapitsa, South Bulgaria.

The soil is classified as Alluvial-Meadow soil according to the Bulgarian classification (KOYNOV, 1987) and as Fluvisol according to FAO classification (FAO, 1997). There is no clear differentiation between soil horizons. The soil profile is AC type: humous upper layer with gradual transition into the parent material. Soil physical properties – soil water permeability, water holding capacity, texture and thickness – strongly influence the amount of water that percolates through the soil profile and the leaching of the chemical elements. This soil has medium to high permeability, high rates of infiltration and hydraulic conductivity (STOICHEV *et al*., 1980). In terms of clay content, two soil layers could be distinguished in the soil profile: 0-30 cm with low clay content (18.6 %) and 30-90 cm with considerably higher clay content (23-27 %). This soil has high bulk density (1.54-1.66 g/cm$^3$) and low water storage capacity (15-22 %).

This Alluvial-Meadow soil could be characterised with slightly acidic reaction, low total nitrogen and humus content and cation exchange capacity, which varies between 20.9-22.4 meq.100 g$^{-1}$.

Soil properties were determined according to the chemical methods for soil analysis by ARINUSHKINA (1970).

Groundwater samples were analyzed for potentiometric pH, $NO_3$-N, $Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$, $HCO_3^-$, and $Cl^-$. Nitrogen analysis in water samples were carried out by direct distillation with 10 % $Fe_2SO_4$ and 0.5% $Ag_2SO_4$ reducing agents (METTODENBUCH, 1955). Potassium and sodium were determined by flame photometer, calcium and magnesium – by atomic absorption spectrometry – AAS (PAGE *et al*., 1982), hydrocarbonate by titration with 0.02n$H_2SO_4$ to pH 4.4 and chloride by Moor method (ARINUSHKINA, 1970). As maximum permitted concentration (MPC) for nitrate, 50 mg.l$^{-1}$ $NO_3$ was used. It is accepted in Europe and Bulgaria as drinking water standard (Council Directive 98/83/EC).

## Results and Discussion

In previous studies (ATANASSOV, 1977; STOICHEV, ATANASSOV and GLOGOV, 1980), a direct connection was clearly established in this soil between the applied N fertilizers and the distribution of nitrate through the whole geological profile under alluvial-meadow soil till the groundwater table. These studies proved the vulnerability of this soil to nitrogen loading and groundwater contamination by nitrate. Nitrate concentration in shallow groundwater strongly depends on the residual soil nitrogen content. Even without fertilizer use there is certain amount of nitrogen available for leaching over the entire geological profile (STOICHEV *et al*., 1980).

Data on groundwater table fluctuation during the monitoring period are presented in Figure 2. The highest water table fluctuation was observed in monitoring well №6 – 140 cm. For the other two monitoring wells, the maximum fluctuation was 110 cm. The dynamics of the groundwater table detected during the monitoring period follows the seasonal distribution of the precipitation.

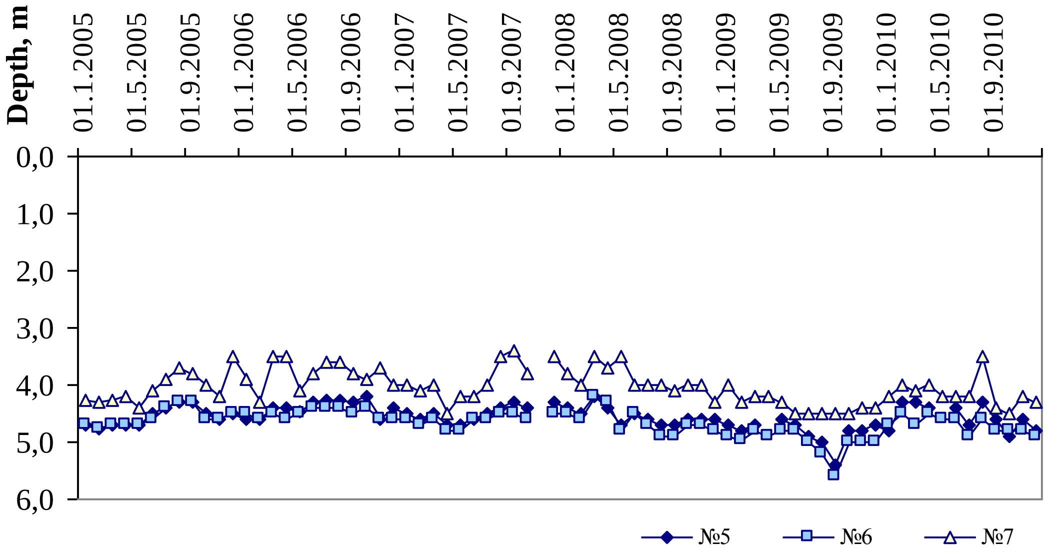
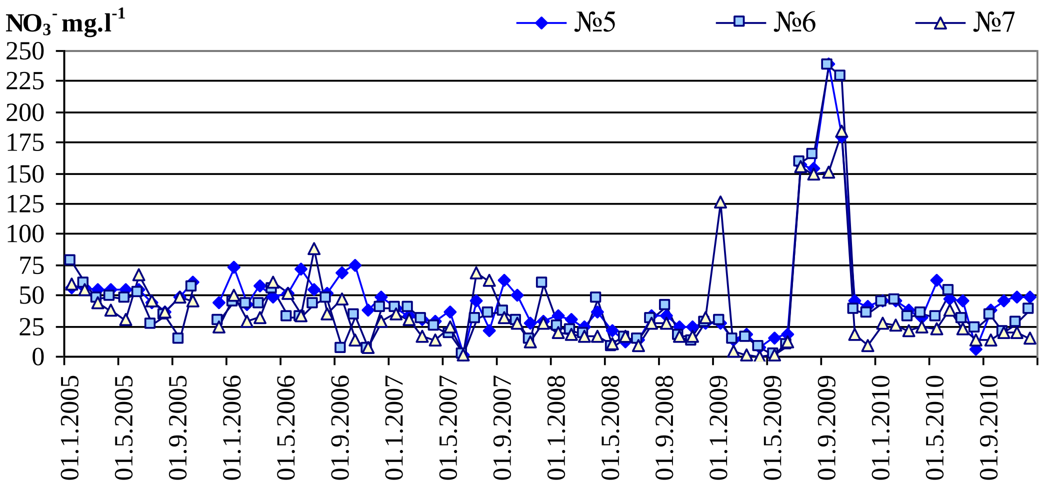
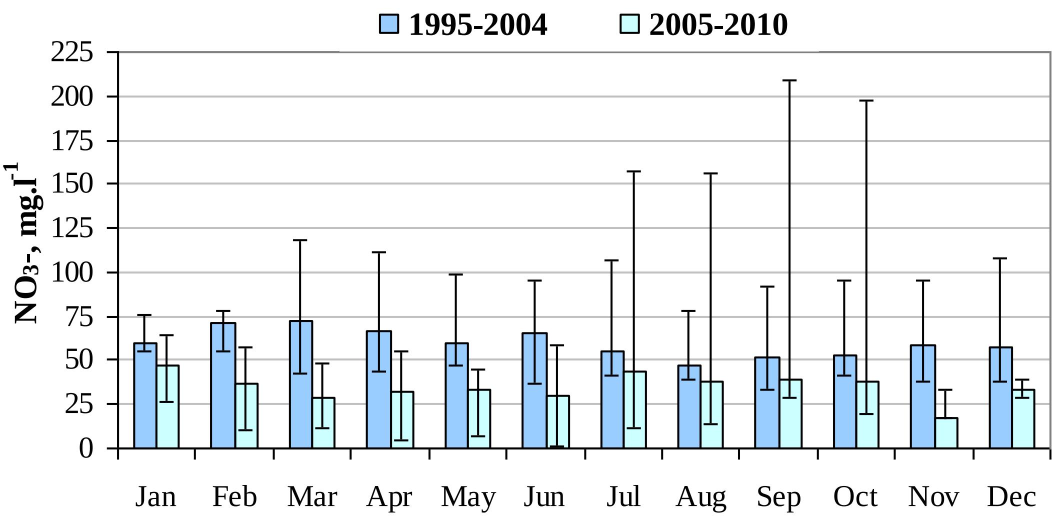



*Figure 2*. Fluctuation of depth of groundwater table (wells № 5,6,7) in the experimental field Tsalapitsa, Plovdiv during the period 2005-2010.

Data obtained in the initial monitoring period 1978 - 1992 (STOICHEV, 1997; SROICHEV *et al.,* 1996) show that nitrate concentration in the shallow groundwater under the experimental area varies in a very large range between 54.9 -171.0 mg.l$^{-1}$ depending on the observation wells (n=8). In almost all cases, the measured nitrate concentrations were higher than the maximum concentration levels for drinking water -50 mg.l$^{-1}$ (Standard for drinking water, 1983). The highest concentrations were detected in the wells situated at the lowest point along the hydraulic slope. The results from the observations made in 1990-1992 years show a significant decrease in the nitrate content in the groundwater under arable lands where fertilizer rates were strongly reduced at rain fed conditions (STOICHEV *et al*., 1996). Data on nitrate content for the periods 1995-2004 and 2005-2010 (Figure 3) confirm the tendency of decreasing nitrate concentration on the average with values around and below the maximum permitted contamination (MPCL).




3a)




3b)

*Figure 3a and 3b.* Nitrate content in groundwater (wells № 5, 6, 7) in the experimental field Tsalapitsa, Plovdiv: a) seasonal dynamics for the period 2005-2010; b) median, minimum, and maximum monthly values for 1995-2004 and 2005-2010

But there are peak values of nitrate during spring-summer-autumn months. The highest concentrations of nitrate in all monitoring wells were registered in July and September 2009 due to heavy rains. Fluctuation in the nitrate concentrations varied in a range of 20-40 mg.l$^{-1}$. There is no statistically proved difference between the three monitoring wells. However, the absolute values in the three wells were different. The lowest nitrate concentration – 37 mg.l$^{-1}$ in average was measured in the samples taken from monitoring well N7 and the highest - 48 mg.l$^{-1}$ in average in well N5.

The groundwater samples obtained from the three monitoring wells have neutral to alkaline reaction in all cases (Figure 4) despite the fact that the soil has neutral to slightly acidic reaction depending on the land uses.

Fluctuation of the pH values in the three wells is almost the same. Two sharp deviations were observed, one in January 2009 when the pH in well N7 (pH 6.5) was almost 1,0 pH unit lower compared to the other wells and the other in July 2009, when the measured pH was 0,8 pH unit higher than in the other two wells. These differences are statistically not significant. No anthropogenic influence was found on the pH values and seasonality was not identified either.

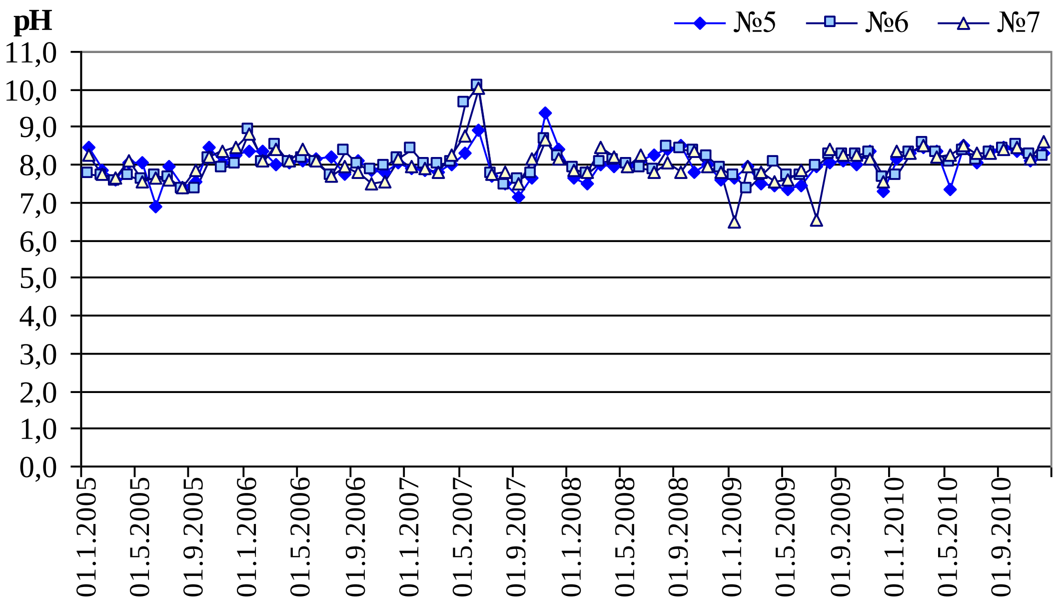
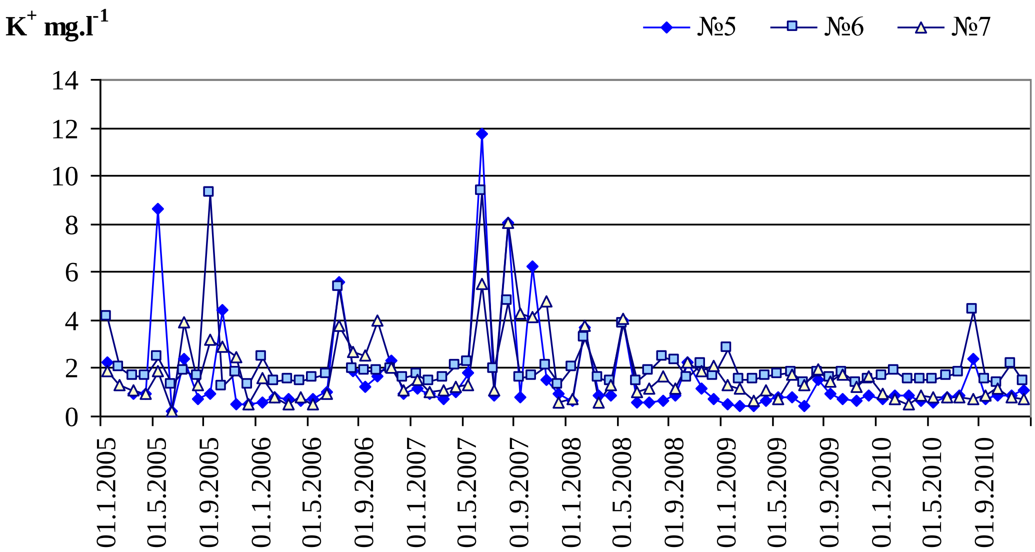


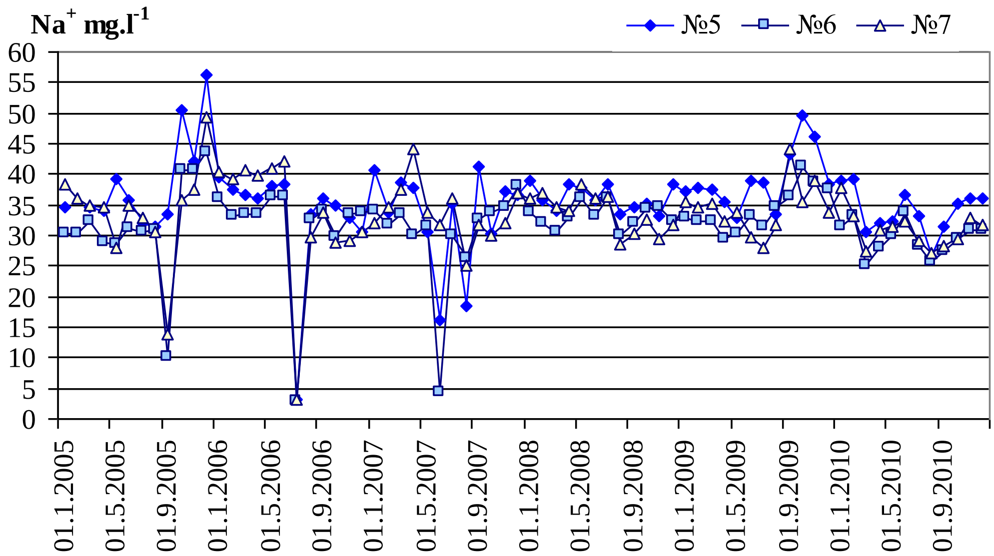

*Figure 4*. pH of groundwater (wells № 5, 6, 7) in the experimental field Tsalapitsa, Plovdiv region during the period 2005-2010.

On Figure 5a, the dynamics in the $K^+$ content in the groundwater is presented. In general, potassium is an element, which is characterized with very low mobility coefficient. It is subject to physico-chemical fixation. The reason for its considerably high content in the groundwater under this watershed is that the coarse soil texture and the high rate of saturation with exchangeable potassium restrict physico-chemical fixation of this element from soil solution during the process of mineralization of organic residues. The highest fluctuation of $K^+$ content was observed in the well N6, which is close to the long-term fertilizer experiment. Potassium content varies from 0.55 to 12.00 mg.l $^{-1}$. Groundwater from this well had the highest $K^+$ concentrations. The only exception was observed during spring months in 2005, when the highest $K^+$ concentrations were measured in well N5 and 6. If we compare the figure for $NO_3^-$ and $K^+$ contents, the same temporal fluctuation can be clearly noticed which indicates that the variation in these elements under this particular soil have one and the same origin.

Dynamics in $Na^+$ content in the groundwater (Figure 5b) seems to have opposite pattern compared with potassium. Sodium content in the groundwater vary in wide ranges (1,00-55,00 mg.l $^{-1}$) Sodium is an element with high geochemical mobility and transitional status in the geochemical cycle and due to this reason it is highly influenced by the changing anthropogenic loads.




5a)




5b)

*Figure 5a and 5b*. Dynamics of potassium (a) and sodium (b) content in groundwater (wells № 5,6,7) in the experimental field Tsalapitsa, Plovdiv region, during the period 2005-2010.

As it is shown on Figure 6, calcium concentrations in groundwater vary in small range, which is not typical for this element due to its high mobility and status in the geochemical cycle of elements. Although the variation is not so big, its temporal pattern seems to follow that of the nitrate content. Calcium concentrations for the study period did not exceed the MPC for drinking water (150 mg.l $^{-1}$) with the only exceptions in September 2005 and July 2010 when the calcium concentration reached 200 mgl $^{-1}$.

The dynamics in $Mg^{2+}$ concentrations in the groundwater is shown on Figure 6. Values for $Mg^{2+}$ content vary from 0.6 to 53.00 mg.l $^{-1}$ and they do not exceed the MPC for drinking water (80 mg.l $^{-1}$). The $Mg^{2+}$ concentrations show very clear seasonal pattern with high values in spring and low in autumn.

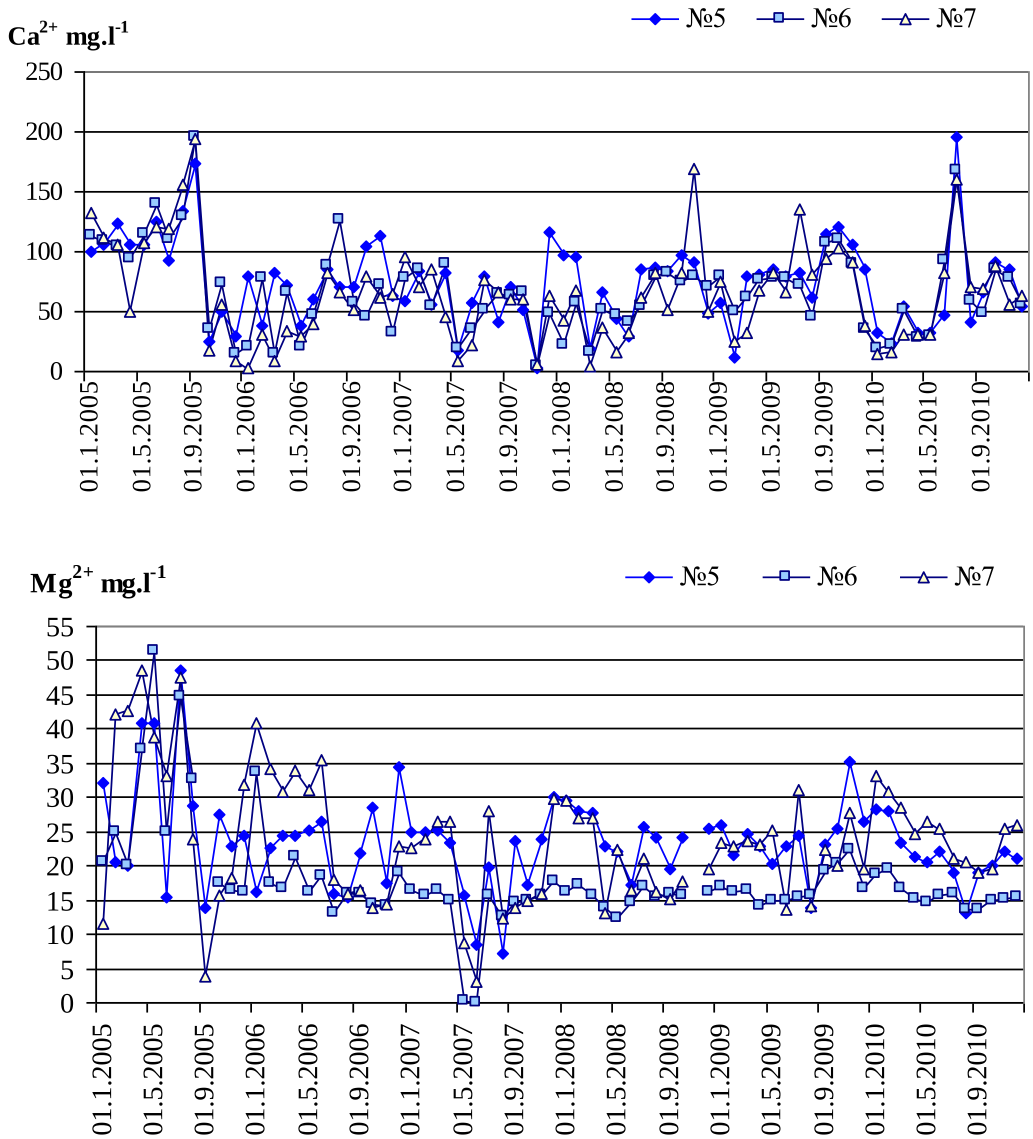
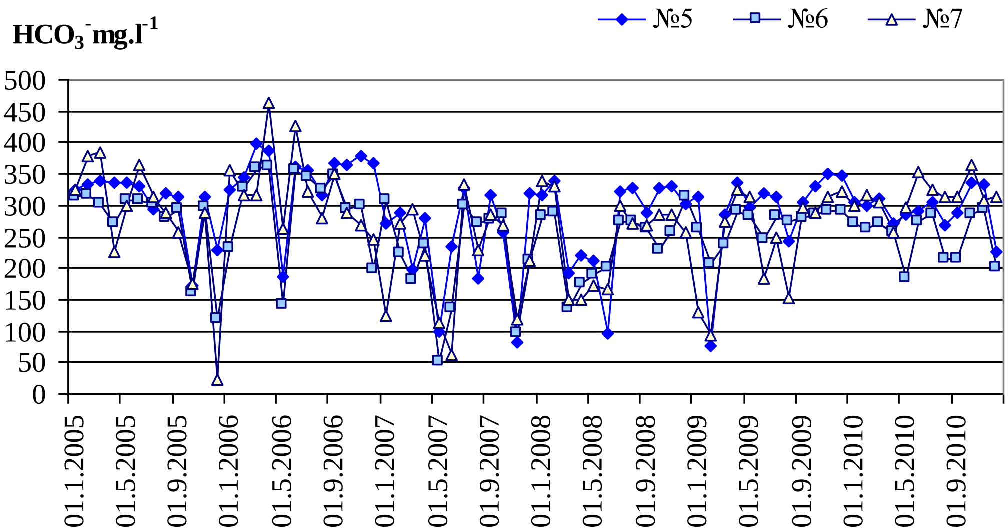



*Figure 6*. Dynamics of calcium and magnesium content in groundwater (wells № 5, 6, 7) in the experimental field Tsalapitsa, Plovdiv region, during the period 1995-2004

Significant variation in the hydro-carbonates content in the groundwater was detected (Figure 7). The data do not show any spatial or temporal dependence from other measured concentrations.

Chlorine content (Figure 7) in the groundwater could be characterised with considerably low variation in space and time. Patterns are almost the same as for calcium concentrations. The data have a narrow range, which is not typical for this element due to its high mobility and in the geochemical cycle. Although the variability is small, its pattern follows that of the nitrate content. Calcium concentrations in the study period did not exceed the MPC for drinking water. The highest $Cl^{-}$ content was observed in July 2006. The anthropogenic loads strengthened the dynamics of chlorine in groundwater because of the absence of mechanisms in the soil to adsorb chlorine from drainage water. Chlorine, like sodium, has a transitional status in the biological turnover of nutrients.



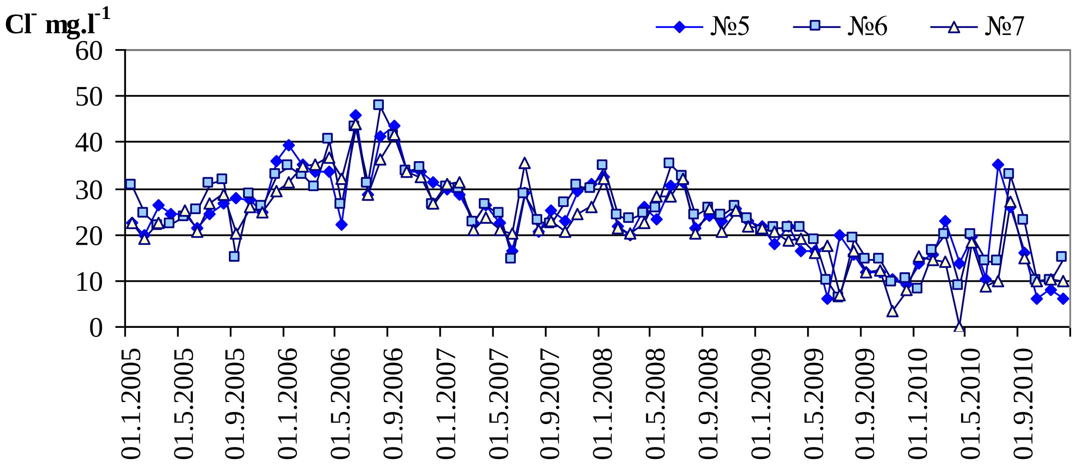




*Figure 7*. Hydro-carbonates and chlorine content in groundwater (wells № 5, 6, 7) in the experimental field Tsalapitsa, Plovdiv

**Conclusion**

Intensive exploitation of alluvial-meadow soil by tillage, fertilizer application, cultivation and irrigation creates conditions for more dynamic water movement through the soil profile and migration of some chemical elements. Groundwater in the study watershed had neutral to alkaline reaction, which did not exceed the maximum permitted pH value of 8.5. Considerably high groundwater table fluctuation was observed during the monitoring period. The highest water table fluctuation was observed in monitoring well №5 – 100 cm. Maximum and minimum levels of the groundwater follow the seasonal distribution of the precipitation.

Nitrate content in the groundwater was influenced by the reduced anthropogenic loads with fertilizers and a decreasing trend in nitrate concentration was observed. Correlation between the dynamics of $NO_3^-$ and $K^+$ contents was recognized which indicate that temporal fluctuation should have the same origin. Sodium content in the groundwater varies within broad limits. Calcium concentrations in groundwater vary in considerably small range and its fluctuation corresponds to that of the nitrate content. Calcium concentrations in groundwater during most of the study period did not exceed the MPC for drinking water. Significant variation in the hydro-carbonates content was observed with slight influence by anthropogenic loads. Chlorine content in the groundwater could be characterised with considerably low variation. It also did not exceed MPC for drinking water.

As a general conclusion, certain effect of anthropogenic loads was observed in this particular soil on the groundwater chemical composition. In order to prevent pollution by agricultural practices, it is necessary to maintain a deficit nitrogen balance. Irrigation has to be done in a way to minimize drainage flow and nitrogen in the irrigation water must be included in the nutrient balance calculations.

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