COST-action 869
Mitigation options for nutrient reductions in surface and groundwaters

WG1: Localisation of critical source areas in catchments (Hamar, Norway May 2007)

1- Introduction
The workshop was aimed at the localisation of critical source areas for nutrient loss in catchments. Critical source areas (CSAs) implicitly include an estimate of nutrient sources and the likelihood of their transport to a given waterbody. However, available approaches to identify these areas vary between those more concerned with sources (e.g. a simple landuse map), those concerned with transport (e.g. the identification of areas where surface runoff occurs) and approaches which explicitly include both. The difference in approaches is driven by the required spatial and temporal scale, the geoclimatic region, the available information and the nutrients and nutrient species in question.

For Water Framework Directive (WFD) purposes, national, regional and large/small catchment studies may be required. These studies may also require that the temporal dynamics of CSAs are estimated: for example the risk associated with different areal extents could be expressed on a return period basis. These dynamics are different across geoclimatic regions, e.g. differences relating to: snowmelt or frozen soil, temporal distribution of annual rainfall or predominant hydrological flow pathways. These phenomena have a large impact on what type of methodology can be practically applied, as does the quality and quantity of data available at each scale.

CSA identification is crucial in the implementation of cost-effective mitigation measures, which, in turn, need to be well proven for a given geoclimactic region (i.e. there is interaction between the identification of CSAs, current land use and the potential effectiveness of mitigation measures – see COST 869, WG3). Difficulties in holistic management of nutrients are compounded by the potential for mitigation of one form of pollution to increase another (*Pollution Swapping*).

As it stands today, few countries have an explicit CSA focus to their nutrient application limitations e.g. the UK has Nitrate Vulnerable Zones; Switzerland has an implicit focus as its law states that localised conditions should be taken into account before application of fertilisers and in Norway CSAs are used by giving high subsidies to use no till on high erosion risk areas. Many countries do have limitations based upon critical periods – e.g. no nutrient applications to frozen soils. Some countries are trying to keep the minimise the source term by regulation with respect to a soil P test (e.g. Ireland and Belgium – avoidance of P saturated areas) and livestock density regulations are more common. These can be nutrient specific and, in some cases, the nitrate directive indirectly regulates P applications.

It is worth noting here that there were a number of comments during the workshop regarding the importance of catchment sinks (i.e. areas where nutrients are likely to be retained), as well as sources.

We have to work within these constraints to best achieve our obligations under the WFD. WG1 delegates have however, identified the following areas which they would like to focus, questions they would like answered and goals they would like to achieve:

1. What methods are available for CSA identification?
    a) A model/tool intercomparison.

2. What are the main similarities and differences between CSAs for P and N (and other pollutants)?

3. Which CSA concepts (and hence methods) are valid for which geoclimatic region?

4. What influence spatial and temporal scale has on how we identify CSAs?
   a) Spatial scale-dependent approaches.
   b) Timing aspects – seasonality of ecological impacts – dynamics of CSAs.

5. What data is required to evaluate adequately our CSA identification methodologies (spatial in particular)? Can we compile a list of existing evaluation datasets within the COST action?

6. What is the potential value of markers/tracers of nutrient sources in terms of spatial information? If markers can identify a pathway how do we quantify its contribution?

7. What is the effect of one mitigation option on another – in terms of modifying CSAs across pollutants? How long do mitigation measures have an effect?

8. Exchange of ideas with other working groups, networking and potentially collaboration.
   a) Links with mitigation options (Joint meeting with WG3).
   b) How does the sensitivity of the ecological target(s) and spatial/temporal variability affect the methodology used to identify CSAs (Joint meting with WG2)? What is the ecological target?
   c) A common EU project.
   d) In Switzerland it is possible to obtain money for COST projects.

9. A hands-on workshop. Involve students in tasks – results into COST action.

10. A consideration of knowledge transfer to farmers and other stakeholders.

11. A dynamic COST 869 website!

2 - Methods for CSA identification?

Although potentially expensive for large areas, direct measurement (e.g. Czech talk: HEZLAR; STAUDINGER) of nutrient concentration in surfacewaters and groundwaters is often a starting point and an option for the general identification of contributing areas. Many countries have long term monitoring datasets which serve this purpose.

Where adequate monitoring data are not available or where a higher resolution of CSAs identification is required, nutrient source, nutrient balance or Index approaches are often used to estimate nutrient pressure. Most of the index approaches have expert/empirical elements or statistical modules and do not take CSA dynamics into account, hence more dynamic approaches may be needed (OLLESCH). CSAs implicitly include source and transport risk. However, some methodologies presented were more focussed and explicit on one or the other (e.g. surface hydrological connectivity is often a primary focus for P and balance approaches for N). The challenge is to identify the best method, given the scale of interest and the characteristics of the area. This led to questions regarding whether or not CSAs are best identified using simple index type approaches or more complex models. The arguments for and against different complexities of model centred around the debate on the difficulties of parameterising and evaluating complex models and the lack of complex and temporally dynamic, flow pathway representation by simplistic approaches. In some cases the opinion was expressed that at very large scales simplistic P Pressure and N balance information may be best as a first approximation: a risk screening.

One problem associated with risk indices is that they are qualitative so thresholds set are partly expert opinion (possibly also a problem of risk being non-linear and simplistic qualitative approaches not able to capture nonlinearities). In a more complex system, the Hungarian presentation (NÉMETH) showed vulnerability mapping of soils for nitrate leaching using GIS layers and deterministic and stochastic models with scale-dependent structures. Cluster analysis was used to create vulnerability classes. Nemeth also looked at a number of simple models and complex dynamic models and questioned the value of the dynamic models as they did not seem to give much better results.

The Australian (COX) experience was an investigation of many different approaches from empirical to physically-based. Models utilised 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.

For all methods presented, one of the main problems is that we have few data to support model evaluation or few data of the right spatial and temporal scale – some of these limitations are technological, that may change with e.g. advances in sensor technology. There is a particular problem with a lack of spatial evaluation data, which means that we can get the right results for the wrong reasons. The German experience (TREPEL) was reported as being limited by data availability, quality and differing approaches in restricting the application of the concept of critical source in practice. This can lead to considerable uncertainty in our estimates of high risk areas which needs to be taken into account when the information is translated to stakeholders. This can be a problem, for example, where the farmers already know the high risk areas better than the scientists' models. When using a model there will always be many assumptions behind what we do: these should be made explicit.

2.1 - Nitrogen-specific methods
For N, an N balance was often identified as the best risk indicator, taking into account atmospheric deposition and losses. Many of the approaches considered were based upon N balances (e.g. the Norwegian N index: BECHMANN). However, the Finnish presentation (RANKINEN) showed that N balance is a fairly good indicator for N leaching in long term but, in the short term, precipitation and cultivation practices (e.g. Manure in autumn, Green manure) were more important.

Romania (DUMITRU) used a more complex approach using data concerning the following characteristics: relief, soil, land use, surface and ground water, obtained yields, number of animals, agricultural management type and water supplying infrastructure.

In Slovakia (ANTAL) Nitrate Vulnerable Zone (NVZ) designation was based on 3 approaches:

- So-called Drastic method (ANTAL) – depth to groundwater, soil properties, topography, properties of the unsaturated zone.
- Field infiltration methods
- Retrospective modelling of piston flow.

The retrospective modelling was used to determine a probability that the piston flow reached a given depth.

2.2 - Phosphorus-specific methods
Many of the methods aimed at CSA identification for P are focuses on the representation of overland flow. In this sense they are weighted towards hydrologically-driven physical transport of particulate-bound P. Hydrology-based approaches are aimed either at the delineation of Variable Source Areas (VSAs: i.e. saturation excess overland flow) or areas where infiltration excess overland flow is likely to occur. The risk of overland flow is then generally combined with a P source term. The approaches used range from static indexes of overland flow potential to dynamic modelling

In Hamar, these approaches included explicit modelling of risk areas using:

- TOPMODEL (COX: where VSAs were targeted for the destratification of topsoil).

- SMDR for surface runoff in a small Swiss catchment (FREY).

- SCIMAP (HEATHWAITE) uses high resolution topographic information and digital terrain analysis to identify areas most likely to be connected to a stream. Climate change effects on this connectivity considered.

- PEDAL (HEATHWAITE) uses a simplistic fuzzy decision tree approach to quantify the likely delivery of P from headwater catchments. A visual assessment of factors which may affect delivery by modifying the connectivity and hence extent of CSAs.

- EUROSEM (a gridded single event model; STRAUSS) to first delineate CSAs and then model the effect of BMPs. They noted that they could not get a good calibration of the location and extent of surface runoff when only using hydrographs and that other features such as tramlines change connectivity (and hence CSAs), but how can these be explicitly modelled?

- PSYCHIC (COLLINS) uses a simplified representation of hydrological flowpaths on a monthly timestep and the Morgan-Morgan-Finney equation for sediment loss. Delivery to the waterbody is estimated as a CDF of distance from watercourse and a particle size selectivity phase for PP.
- There was a more top-down approach described by STAMM (Lazzarotto, 2005) where the hydrograph was split into fast and slow response (BFI type system) to generate a transport risk map which is overlain with the source risk map.

These approaches often require a relatively high resolution DTM and are generally only evaluated by catchment outlet hydrographs – it was discussed that the lack of spatial data on surface runoff/soil moisture etc. is a significant problem.

Other, more simplistic representations of hydrological pathways were presented.

- Models can be derived using a soils map and then a system for determining flow partitioning (e.g. STAMM, using an expert risk classification and decision-tree approach) or (e.g. HOST BFI classification used in many UK models such as PIT (Heathwaite *et al*. 2003) and PSYCHIC (Davison *et al*., in press).
- Many of these approaches have an explicit estimate of connectivity of a soil unit to the surface waterbody using variables such as: topography, distance from stream and presence of artificial drainage. These are the type of variables also used in index-type approaches such as the Norwegian and Danish P index. Part of the difficulty in implementing these simplistic representations of nutrient transport is the assignment of the relative weighting given to a particular variable.
- TREVISIAN concluded that surface runoff, spatial organization of crops, fertilization regimes and the general structure of the rural landscape determined the location of CSAs.
- COX presented a simplistic P mobility index which was based on P adsorption, CEC, saturated hydraulic conductivity and macroporosity.
- HACIN showed a land use and soil type approach of looking at the P problem, highlighting the differences of an approach aimed at SRP losses compared to TP/PP losses.

There was a discussion on whether or not there is too much focus on surface pathways.

- Should we focus more on throughflow and groundwater?
  How are macropores dealt with in models and how do we get good information on presence/extent/connectivity of macropores?
- Rarely explicitly included in models and often just a bypass routine.
- There were also similar discussions regarding artificial drainage.

- It is difficult to know the extent and operational status of drains.

2.3 Nutrient Source/agronomic practice-based approaches
Some of the methodologies presented did not explicitly include hydrology-driven nutrient loss; they were more concerned with the potential nutrient source and likelihood of mobilisation.

- A methodology which looked purely at delineation of critical phosphorus based upon 'Phosphate Saturation Degree' (PSD - 40% reduced to 35%) was presented by SALOMEZ for Flanders. The combined effect of extremely variable data and a requirement of high certainty of predicted values can lead to the underestimation of the size of P-saturated zones. Although a stricter PSD limit is a crucial factor in reducing P-losses this research revealed that the parameters in decision making need to be harmonized with data properties for an efficient delineation of risk zones.

- The potential for P-loss using soil samples from the National Soil Archive of Scotland was presented by SHAND. The methodology was aimed at identification of "change point" values which could then be mapped.

3 - Similarities and differences between CSAs for P and N.
Phosphorus and Nitrogen partly behave in different was in the environment as do different P and N species. This can lead to different methods being required to identify CSA and to potentially conflictory results of mitigation methods.

1) Nitrate (NVZ's)
2) Dissolved P (entire flow pathway)
3) Particulate P (well-connected VSA's and preferential flow pathways).

Many of the N focussed talks considered N balance and N leaching studies as methods to identify CSAs for N (NVZs).

Whereas the Norwegian P index was based on a source, transport and delivery approach, whereas the Norwegian N index was based on N balance – N leaching. Results showed that losses of P and N in the case catchment have different critical source areas and pathways of loss.

4 - Geoclimatic variations in CSAs.
From a COST initiative point of view the different geo-climatic regions of Europe may have their own problems and challenges, but they will also have things in common and the diverse experiences of the participants should help to identify the most appropriate solutions to identify CSAs and which BMPs are most effective for different geo-climatic regions?

Of course, these may vary more within member countries than between – e.g. Greece was divided into 3 geo-climatic regions in the Skoulikidis presentation. There may be variability resulting from differences in climate, landscape, land use, land management, and socio-economic and political factors. The factors that affect our approaches in different geoclimatic regions include:

- Scandinavian and alpine regions need to take account of the cold climate (snow cover, freezing, frozen soil and the subsequent spring snowmelt).

- The BECHMANN presentation showed that freeze-thaw cycles can release more P and need to be accounted for in their P index

- In some countries such as Norway and Finland the primary receptors for nutrients are lakes whereas in others (e.g. the UK rivers are the primary receptors). The residence times of the lakes become important – link with WG2. Climatic differences affect the temporal pattern of rainfall and whether or not it coincides with the growing season. This was highlighted by an example in the discussion where in the UK there is a debate on the importance of the high flux of nutrients through rivers during the winter when plants are less active.

- Differences in rainfall intensity, and the temporal distribution of annual rainfall together with differences in soil properties control whether or not we need to consider hortonian or saturation excess overland flow (or both).

- There are large differences in the proportion of each country that is used for agriculture which may limit the options for dealing with CSAs. E.g. 20% of Slovak Republic could be designated as a NVZ, it is not possible to remove agriculture from this area.

- The Spanish presentation (DELGADO) showed how irrigation and water use efficiency, rather than rainfall, can be a key factor determining P losses.

- In many countries the presence and condition of artificial land drains is a key factor.

5 - Scale Dependencies and Complexities

Scientifically we need to understand cause-effect relationships but environmental variability makes risk management at catchment and basin scale problematic. Thus we need tools and practices to enhance sustainable practices and document the effects of implementations (i.e. ongoing monitoring). The challenge is partly a matter of scale and of technology – we may not have the tools to do what we would like to do. In some cases we need to “translate” the knowledge obtained at plot, field and small catchment scales to larger catchments. Moreover, in general, decisions are made at field scale, but high loss risk areas can be at sub-field scale.

Scale dependent approaches – often large scale approaches use relatively simple P indexing systems which are often based upon the Pennsylvanian P index, but modified to describe local conditions (e.g. Norwegian P index and Danish P index). These approaches describe a relative static risk – no dynamics are simulated.

The Dutch presentation (VANDERSALM) showed a scale-dependent approach with a national scale dynamic mechanistic model (STONE, Wolf et al. 2005). A metamodel of STONE was created for regional scale studies and a simplistic mechanistic model (PLEASE) for local/field scale.

The German presentation (Trepel) described scale-dependent modelling approaches: some models were for the whole country, down to detailed models that simulate the dynamics of source areas (VSAs) in small catchments. They advocated a 2 tier approach where a risk screening is carried out then local consultants use more sophisticated models to give detailed risk estimates.

Some studies showed that, even in plot studies, systems such as set aside are not always consistent with their low risk status?

In general we may agree about the likelihood of a fractionation of P species in certain types of flowpathway, but quantitative modelling of these phenomena is weak.

Should we include the critical times, as well as areas, (and sensitivity) for the receiving water body? The discussion sessions seemed to suggest we should.

6 - Mitigation Options

Mitigation options implicitly change the status of land units and hence their CSA status. For example, the effect of different soil types and tillage practices on soil structure and how flow pathways were altered was considered by TREVISIAN who found that for cultivated fields managed by traditional dairy systems, tillage practices tended to increase soil crusting susceptibility and consequently to increase the frequency of surface runoff and erosion, whereas The risk was significantly lower for the fields cultivated by more intensive farms equipped with a better machinery; a ranking of farming practices for their effect on overland flow was also presented.

In The Swedish study (*ULÉN*) presented the critical Agricultural practices were shown to be slurry application to wet soil, ploughing in the ley, poor fertilizer incorporation and incorporation of green manure. The P losses increased in order for:

- dairy with grass
- dairy with lucerne
- monoculture with barley
- organic farming with cattle slurry
- stockless organic farming with green manure

In Norway 35-40% of arable land is conservation tillage and the Norwegian experience is that it usually reduces nitrate losses and a significant reduction of TP loss overall - even if it changes the fractionation. But it means that more herbicides are needed. There was a comment, during discussions, that conservation tillage increases the N loss to the atmosphere.

The Austrian (STRAUSS) exploration of cost-effectiveness of change to pasture, winter cropping and mulching showed examples for a catchment and asked the questions: how general are these results? - are variations due to mainly hydrological characteristics? The efficacy of mitigation options is scale-dependent – may not be as good as thought as not all P reduction shown may have reached the stream. There is a need to derive information on the optimal combinations of mitigation options in terms of applicability and cost effectiveness – but these often have been shown at plot scales – do they hold at larger scales?

HACIN (Slovenia) looked at the effect of wetlands in modifying nutrient loadings and referred to a study which showed variable results - some showed an increase and some a reduction in nutrients. He linked the P losses to the instream effect of uptake by macrophytes; this is an area which is seldom linked but has implications for cost effectiveness of mitigation measures as different waterbodies have different sensitivities to P reductions.

Many of the studies highlighted concerns over pollution swapping.

- In Australia, the mitigation technique of using Ca additions to the soil had the impact of also changing the soil structure and flow pathways and hence knock-on effects.

- Often, there is a shift of P forms with erosion reduction - Do BMP‘s for different threats disturb each other (environmentally and economically)?

- A Swedish study showed low N/P ratios in drainwater and discussed the importance with respect to lake cyanobacteria concentrations - a balanced reduction of P and N is therefore required. (Link to WG2).

- HACIN also looked at the effect of wetlands in modifying nutrient loadings and referred to a study (Hydrology and Earth System Sciences (2004) 8: 673–685 Wetland nutrient removal: a review of the evidence - J. Fisher and M.C. Acreman), which showed variable results some showed an increase and some a reduction in nutrients – partly down to sampling regime.

Perhaps some of these issues can best be addressed with links to WG3.

7 - Climate Change
Climate change was mentioned a few times, notably:

HEATHWAITE showed the potential increase in VSAs estimated under climate change.

ULÉN showed an increase in summer lake temps in Sweden and discussed changes in lake sensitivity and length of critical period.

8 - Markers/Tracers
FREY showed that Herbicides could be used as tracers from different fields where correlation between connected catchment area and herbicide loss was found.

LITAOR used sulphate as a tracer for P, as it was sourced from a particular place in catchment. Re-flooding of the wetland created reducing conditions which released huge amounts of S and P.