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The EU Water Framework Directive requires Member States to achieve "good ecological quality" in (most) surface waters by 2015. This EU Workshop began to define what this means with respect to eutrophication. The definition will also be relevant for the Urban Waste Water and the Nitrates Directives, and for OSPAR.

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Germany, Sweden

National objectives for P-recovery announced in Germany and Sweden

Both Germany and Sweden have announced national objectives for phosphorus recovery for recycling from sewage. Sweden's action plan centres on recycling P to land through sewage sludge use in farming, whereas the German Federal Environment Ministry (UBA) suggests recovery for recycling in sewage works.

The German Federal Environment Ministry (Umwelbundesamt UBA) announced in March 2003 the objective of developing phosphorus recovery for recycling from sewage and other wastes. The press release followed the organisation of a symposium (Berlin, 6-7 Feb. 2003) which brought together around one hundred German experts, regulators and water industry operators to discuss P-recovery from sewage and wastes. It also followed a call by the Baden Württemberg Land environment Minister in February to ban sewage sludge spreading on farmland.

UBA suggests that existing taxes on waste water could be used to support the technical development of phosphorus recycling.

Sweden has also announced the objective of developing phosphorus recycling, but **the Swedish EPA (Environmental Protection Agency) action plan published in December 2002 proposes to achieve 60% P-recycling by 2015 largely by re-use in agricultural by sewage sludge spreading.** The Agency estimates that at present 46% of sewage phosphorus is already effectively recycled through agricultural sludge application. Sweden hopes to move back to agricultural sludge spreading following severe restrictions imposed by farmers over the last few years, because of concerns about health or pollution contamination risks, with the introduction of new EU sludge spreading quality criteria which are expected to be proposed soon by the EU Commission (see below).

Both the Swedish and German authorities recognise that phosphorus can be recovered for recycling by various process including recovery from wastewaters in sewage works or from sewage sludge incineration ashes.

German Environment Ministry press release (in German):
<http://www.umweltbundesamt.de/uba-info-presse/presse-informationen/pd01903.htm>

Translation shortly available at:
<http://www.nhm.ac.uk/mineralogy/P-recovery/>

Swedish EPA press release (in English):
<http://www.internat.naturvardsverket.se/index.php3?main=/documents/press/2002/p021216.htm>

EU Commission (DG Environment) proposed new rules concerning agricultural spreading of sewage sludges:
<http://europa.eu.int/comm/environment/waste/sludge/index.htm>

Water treatment

Sewage sludge management in Europe

Large quantities of Europe's sewage sludge production are currently recycled by agricultural spreading, effectively allowing both economic disposal and recycling of the nutrients (phosphate and nitrogen) and organic contents¹. However, this route is under significant pressure because of public concerns – objectively justified or otherwise – concerning risks of heavy metal, pathogen or pharmaceutical chemicals contamination of soils and/or of agricultural products (food). There is also concern about risks of run-off of such contaminants to ground and surface waters (drinking water supplies).

1: approximately 36.4% of European sewage sludge is spread on land, according to Rebecca Renner in "Sewage Sludge, Pros and Cons", Environmental Science & Technology vol. 34 - I.19 1, October 2000, text at: www.mindfully.org/Pesticide/Sewage-Sludge-Pros-Cons.htm

These concerns have led in some countries (eg. UK) to agreements between farmers organisations, water companies and supermarkets to limit levels of contaminants and to ensure pre-treatment of sewage sludges before spreading, or in other countries (eg. Sweden, Switzerland) to farmers organisations recommending or legal implementation of bans on agricultural sludge use.

This EU report analyses the different options for sewage sludge disposal and concludes that agricultural use is the best environmental and economic solution. The development of the agricultural spreading route requires the improvement of both sludge qualities and of public confidence. **Sludge quality guarantee and labelling systems** will be a key tool to improve this, but with a

significant cost to water companies. Research into sludge quality, pollutants and their fate in the environment and in food, and the publication of this research will be important in facilitating public acceptance.

Proposed tighter regulations

In order to develop public confidence, the EU is proposing a significant tightening of regulations concerning biosolids spreading (currently covered by *EU Directive 86/278*), to require fuller sludge pre-treatment before spreading, to lower soil and sludge limits for heavy metals, and to introduce limits for certain organic contaminants.

The report estimates that this will have little impact in countries where national sludge spreading regulations are already much more stringent than the existing EU Directive requirements (Denmark, Finland, Sweden, Netherlands). **Implementation costs will however be significant** in countries where existing national regulations are only somewhat more stringent (Austria, Belgium, France, Germany, Poland) or are similar (Greece, Ireland, Italy, Luxembourg, Portugal, Spain, UK, Estonia, Latvia) to the current EU Directive.

Who will bear the costs ?

The total cost implied by the proposed Directive tightening for the 15 EU Member States is estimated at almost **1 billion Euros/year**, with a large proportion of this cost being born by four EU states who currently recycle to agriculture large volumes of sludge (Germany 29% of total cost, UK 26%, France 16%, Spain 11%).

As a European average, sewage sludge disposal costs are estimated at 318 €/tonne dry matter (DM), that is at 0.12 €/m³ of drinking water supplied, compared to total sewage costs of 0.87 €/m³, that is 14%. Total drinking water service costs (production and treatment) are estimated at 2.2 €/m³. Average sludge disposal costs are estimated to vary from 263 – 344 €/tonne DM. Landspreading of solid/semi-solid sludge is estimated to cost 110-160 €/tonne DM, of which 10-30% of the cost is covered by the fertiliser value of the sludge.

The report estimates that implementation of the proposed sludge spreading Directive tightening would result in 67% (short term) rising to 83% (long term) of sewage sludge failing limit values for heavy metals or organic pollutants in the absence

of an effective “pollution prevention” policy to reduce the input of such contaminants to raw sewage. Such a policy could reduce the proportion of sludge failing limit values to around 25%. Despite this, the total cost to all stakeholders (water companies, farmers, industry, public / consumers, local authorities) of the new Directive *with* the pollution prevention policy could be around 15% higher than *without* such a policy.

A key difference is that *without* the pollution prevention policy, 60% of the costs are born by local authorities (ending up on the public’s water bill), 20% by farmers and 16% by the general public (environmental and health impacts), whereas *with* the policy 60% of the costs are born by the pollution producing industries (discharge pollution prevention costs).

“Disposal and recycling routes for sewage sludge”, report for the EU Commission – DG Environment – B/2, SEDE/ Arthur Andersen, January 2002. Available at: http://europa.eu.int/comm/environment/sludge/sludge_disposal.htm

Phosphorus recovery

Germany

Struvite shows to be good fertiliser

Struvite precipitation was tested in a pilot animal slurry treatment plant in Northrhine-Westfalia, Germany, sponsored by a farmers’ cooperative. The recovered struvite was then tested for fertiliser potential (nutrient availability to plants) using randomised pot experiments with four replicates and oats (*Avena sativa*) and ryegrass (*Lolium multiflorum*).

The pot trials used luvisol (pH-CaCl₂ 6.5) and a 1:1 brown earth / quartz-sand mix (pH-CaCl₂ 5.5) for the phosphorus availability experiments, and brown earth (pH-CaCl₂ 5.5) for the magnesium and nitrogen availability experiments.

Performance of struvite was compared with that of raw phosphate and dicalcium phosphate (DCP), all at 0.6 g P₂O₅/9 kg pot; with NH₄NO₃ + CaSO₄ at 0.4 gN/ 9 kg pot ; and with MgSO₄ (KS) at 1 mg Mg/9 kg pot. Phosphorus uptake by

oats was significantly higher (2.5-3x) for each of the three P applications (struvite, raw phosphate, DCP, compared to the no P addition control), but slightly lower with raw phosphate than for struvite and for DCP (soil pH 5.5).

Ryegrass also showed considerably higher P uptakes for the three P treatments, but with P uptake in the raw phosphate treatment significantly lower than for struvite and DCP (soil pH 6.5).

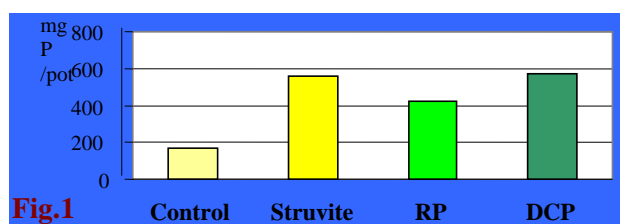


Fig.1

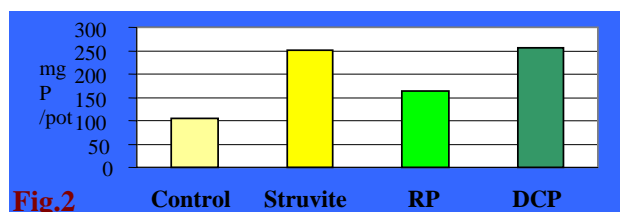


Fig.2

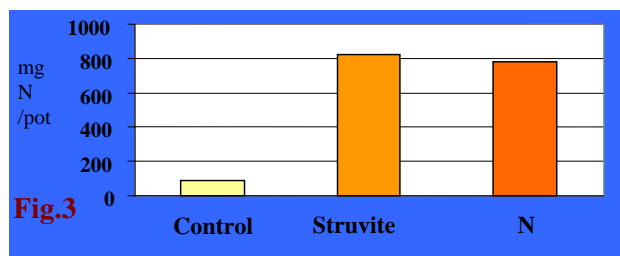


Fig.3

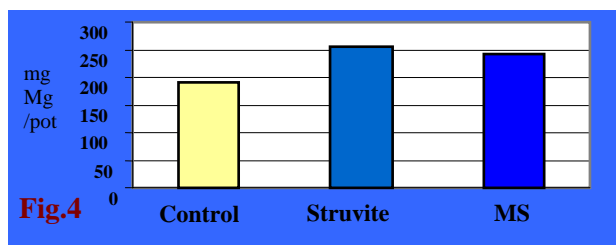


Fig.4

Fig.1: P uptake of ryegrass, soil pH 5.5,
Fig.2: P uptake of ryegrass, soil pH 6.5
Fig.3: N uptake of ryegrass, soil pH 5.5
Fig.4: Mg uptake of ryegrass, soil pH 5.5

This suggests that struvite P availability is as good as commercial fertiliser at soil pH 5.5 and 6.5 (whereas raw phosphate effectively showed a lower P availability at the higher soil pH).

Nitrogen uptake was significantly better with struvite than with ammonium nitrate for oats (both at soil pH 5.5), with both being considerably higher (6-9x) than for the control.

Magnesium uptake also marginally better from struvite (oats, soil pH 5.5) than with KS, with both being somewhat higher than the control (1.3–1.4x).

Struvite precipitation was tested in a pilot animal slurry treatment plant in Northrhine-Westfalia, Germany, sponsored by a farmers' cooperative. The recovered struvite was then tested for fertiliser potential (nutrient availability to plants) using randomised pot experiments with four replicates and oats (*Avena sativa*) and ryegrass (*Lolium multiflorum*).

The authors conclude that nutrient availability in struvite is at least comparable to that of commercial fertilisers, even on near neutral soils, and that struvite precipitation is thus a useful tool for separating phosphorus and nitrogen from animal slurry in a form which can be used as a fertiliser.

“Plant availability of phosphorus, nitrogen and magnesium applied with magnesium-ammonium-phosphate (struvite) derived from animal slurry”. Poster presented at the IWA/NVA Conference “From nutrient removal to nutrient recovery”, Amsterdam, 2-4 October 2002. Poster summary available at

<http://www.nhm.ac.uk/mineralogy/P-recovery/>

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Piggery wastes

Struvite formation for N-removal

The authors studied the results of adding a struvite (magnesium ammonium phosphate, MAP) formation pre-treatment step to a 5 / 10 litre laboratory scale installation treating scrapper-type piggery waste waters. The water treated had a relatively low carbon/nitrogen ratio because most solids are separated out and not discharged in the liquid: COD around 4,300 mg/l, total nitrogen around 1,700 mgN/l, NH₃-N around 1,600 mgN/l.

The pilot treatment plant used a 5.4 litre upflow anaerobic sludge bed reactor (AUBF) with a hydraulic residence time of 2.7 days followed by a 10 litre aerobic reactor with a hydraulic residence of 5 days, with sludge recycling. When the system was operated without the struvite formation pre-treatment step, the effluent nitrogen concentration ranged from 500-1,000 mg/l.

Struvite formation was induced by adding phosphate and magnesium ions to the influent before it entered the AUBF reactor, to concentrations equivalent to 1:1 ratio to 1,000 mgNH₃-N/l (0.71 molar), and increasing pH to 8.5. This caused ammonium to be precipitated as struvite, increasing the inflow carbon/nitrogen ratio from 2.8 to 6.0.

The struvite formation caused inflow organic carbon to be reduced by only 13% (average influent TCOD reduced from 4,750 to 4,140 mg/l). The struvite pre-treatment however reduced the process effluent total nitrogen concentration from 790 – 1,605 mgN/l (43% nitrogen removal only) to 350 mg total N/l (**80% nitrogen removal**).

The authors conclude that struvite precipitation (by phosphate and magnesium addition) can be an effective process for improving nitrogen removal from piggery waste waters with low carbon/nitrogen ratios. They suggest that the struvite could be recovered and recycled as a fertiliser.

“Struvite formation and subsequent biological process for nitrogen removal in the piggery wastewater”. Paper at the IWA/NVA Conference “From nutrient removal to nutrient recovery”, Amsterdam, 2-4 October 2002.

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Japan

P-recovery from piggery wastewaters

Aeration followed by settling was tested as a phosphate precipitation process for screened (1.5 mm mesh) swinery wastewaters from the Japan National Institute of Livestock and Grassland Science (NILGS) piggery, Ibaraki, Japan.

Initial batch experiments were carried out to assess feasibility and operating parameters, then a continuous-mode pilot plant (0.6m diameter, 2.43m high) was tested over a 50-day run. The swine wastewater used was relatively variable, depending on washing of the installations, with average soluble concentrations of P-PO₄ = 5.2 mmol (161 mgP/l), N-NH₄ = 39 mmol (543 mgN/l), magnesium 4.9 mmol (119 mg/l) and calcium 3.4 mmol (136 mg/l).

The initial batch mode experiments used 30 litres of screened piggery wastewater. This was first aerated in a 21cm diameter column, using a ceramic diffuser for aeration, for up to 3 hours. An optimal aeration rate of around 13 m³air/m²/hour was found, enabling the wastewater pH to be raised from 7.0 to 8.0 in half an hour, and up to pH 8.5 after 3 hours. This pH increase corresponded to a considerably decrease in I-C concentration in the wastewater, indicative of dissolved CO₂. An aeration rate of half this level was found insufficient to raise pH (no phosphate precipitation) whereas a doubled aeration rate allowed a slightly higher pH increase (pH 8.7) but without bringing a significant change in phosphate precipitation.

2 hours aeration was found to be adequate to reduce both soluble phosphate and magnesium from around 5 mmol to below 1 mmol, with continuing aeration on to 3 hours resulting in only a very small further reduction. The changes in concentrations of soluble/insoluble phosphate, magnesium, ammonium and calcium led to the conclusion that a mixture of struvite (magnesium ammonium phosphate) and calcium phosphate crystals were being formed.

To test the settling of the precipitated phosphates, half of the batch aeration solutions (15 litres) was poured into a vertical 10cm diameter cylinder (190cm depth), with sampling ports along its sides, and allowed to settle for one hour. Samples were taken from ports 40 and 90 cm above the cylinder bottom

after 15, 30, 45 and 60 minutes, and from the bottom of the cylinder at the end of the hour settling.

Sampling showed that after 15 minutes, the phosphate crystals had almost entirely settled below the port 90cm above the cylinder bottom, but that 30 minutes were required to reach low levels 40cm above the bottom. After one hour, for different batches, 90-92% of the precipitated phosphates (calculated from changes in soluble concentrations during aeration) were recovered in the settled solids on the cylinder bottom.

Continuous pilot plant

On the basis of the information gathered from the batch experiments, a **pilot reactor** was constructed an aeration zone (central 0.3m diameter column) surrounded by a settling zone (2.43m high, 0.6m diameter outer cylinder). The aeration rate was set at 26 m³air/m²/hour, higher than the optimum derived from the batch experiments, to allow for scale-up efficiency loss. Total residence time in the whole reactor was 22 hours (inflow/outflow rate = 0.024m³/hour of wastewater), with an upflow velocity of 0.11 m/h in the settling zone.

This installation achieved a pH of 8.0 in the aeration zone, but this decreased to 7.6 at the reactor outlet (at the top of the settling zone), probably because the wastewaters were becoming anaerobic in the unmixed settling zone.

The plant, over a 50-day run, achieved average removal rates of 65% for soluble phosphate, 51% for magnesium and 34% for calcium. The plant achieved a reliable concentration of 1.7 mmol soluble phosphate despite very variable inflow concentrations (average 4.8 mmol P-PO₄ inflow wastewater).

Around ¾ of the precipitate was calculated to be struvite, and 1/3 calcium phosphate. The authors considered that the settled sludge, containing a mixture of small suspended solids from the screened wastewater and the precipitated phosphates, could be dehydrated to provide a soil conditioner or used in compost, because of the fertiliser value of the struvite.

The authors suggest that the process provides a **simple and reliable method of consistently reducing phosphate and magnesium levels in piggery wastewaters**, both reducing risks of struvite scaling and deposit problems and allowing phosphate recovery for recycling to agriculture.

“Removal of phosphate, magnesium and calcium from swine wastewater through crystallization enhanced by aeration”. *Water Research* 36, pages 2991-2998, 2002.

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Tennessee

Optimising struvite recovery from swine wastes

Waste waters from two different piggeries were used for laboratory struvite precipitation experiments, to better define optimal conditions for struvite precipitation, as a means of reducing waste water phosphate content and to recover phosphates in a recyclable form.

High concentration supernatant was collected from a pull-plug pit under a pig farrowing unit (average around 1,000 mg/l phosphate, 51,000 mg/l COD). Low concentration supernatant was taken from a holding pond at a feeder pig unit operating a recycle flush system (average 230 mg/l phosphate, 410 mg/l COD).

Previous work by the authors had shown that a 90% reduction in soluble phosphate content could be achieved by adding magnesium to a 140,000 litre pig slurry holding pond, resulting in struvite precipitation. This work aimed to better define cost-effective conditions for struvite precipitation, and to try to achieve precipitation in a recoverable form, allowing separation from the slurry and recycling. (In “Environmental Technology” 22, pages 1273-1278, 2001 available at [http://www.nhm.ac.uk/mineralogy/phos/Nordwijkerhout/Environmental Technology/Burns.doc](http://www.nhm.ac.uk/mineralogy/phos/Nordwijkerhout/Environmental%20Technology/Burns.doc))

Experiments used 200 or 400 ml of waste water in a laboratory beaker, with addition of 60% magnesium chloride as a magnesium source, and in some cases sodium hydroxide to raise the pH. Magnesium chloride has the advantage of high solubility (ease of handling, effective reaction) but is slightly acidic, tending to reduce the pH of the wastewater when added, which is contradictory with the fact that struvite precipitation is facilitated by a pH increase.

Different reaction times of 10, 20, 30 and 40 minutes were tested, both at the wastewater’s initial pH 7.4 and 7.5) and with pH adjustment to pH 8.5. A

reaction time of 10 minutes proved to be adequate, with an improvement in phosphate precipitation of only 1-4% when after 40 minutes. The pH increase, on the other hand, significantly decreased the residual phosphate concentration by a factor of 2-3. However, the authors note that the use of sodium hydroxide to adjust pH is unlikely to be economically viable.

Molar ratio

The effects of increasing the molar ratio of added magnesium were also tested. The authors' previous work cited above had suggested that the molar ratio of magnesium addition should be based on soluble phosphate and soluble magnesium (not total) with ratios significantly higher than 1:1 being appropriate. In this work, ratios varying from 1.6:1 to 3.5:1 were tested.

The results showed that the ratios higher than 1.6:1 soluble magnesium:soluble phosphate did not significantly improve phosphate precipitation. Further experiments would be needed to evaluate whether adequate precipitation efficiency could be achieved with ratios lower than 1.6:1 (the theoretical molar ratio of magnesium:phosphate in struvite is 1:1).

Seeding

Experiments were also made with the addition of struvite or of sand particles to attempt to "seed" the precipitation reaction and generate larger struvite particles, easier to separate from the waste water, recover and recycle.

The addition of struvite seed did not appear to usefully increase the size of precipitated struvite particles, nor to improve the phosphate precipitation. **The sand seed, however did result in an increase in precipitated struvite size**, after 1-2 hours, though the larger particles appeared by observation to be independent of the sand particles. Also the phosphorus removal was improved when the sand seed was added. However, the improvements gained were not significant enough to justify additional costs or manipulations. The authors suggest that more work is needed in this area.

The precipitated phosphate was analysed and showed the presence of both struvite (magnesium ammonium phosphate) and brushite (a calcium phosphate). The molar ratio of magnesium: ammonium: phosphate in the precipitate was 1:0.74:3.2, showing that it is not pure struvite, but has a **higher phosphate content**

than struvite which would be positive for recycling. This also suggests that the molar ratio of magnesium addition could probably be reduced, thus reducing costs.

The authors concluded that further experimental work is needed on a number of parameters, and also the development of on-farm processes producing precipitated struvite in a form which can readily be recovered and recycled.

"Optimization of phosphorus precipitation from swine manure slurries to enhance recovery", paper presented at the IWA/NVA International Conference "From nutrient removal to nutrient recovery", Amsterdam, 2-4 October 2002, publication expected in Water Science and Technology <http://www.iwaponline.com/wst/toc.htm>

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Struvite

Parameters for struvite deposits and recovery

The two papers on struvite precipitation chemistry and behaviour published by Doyle, Parsons *et al.* in "Water Research" provide a thorough summary of current knowledge and references in this area, and further new experimental evidence of the differences in struvite precipitation rates on different materials at different roughnesses.

The overview paper "*Struvite formation, control and recovery*" provides information on the impacts of struvite deposit formation in sewage works, struvite chemistry, precipitation and scale formation factors, struvite deposit prevention techniques, and struvite recovery for phosphate recycling.

Struvite deposits are an increasing problem in sewage works worldwide, in particular as a consequence of nutrient removal installation and of efforts to reduce sludge production, which both concentrate nutrients in sludge streams (and thus ammonia, phosphate and magnesium, which can react to form struvite – magnesium ammonium phosphate). Struvite deposits can reduce pipe diameters, increasing pumping costs or even

necessitating acid soaking to dissolve the deposits or pipe replacement, and can also affect filters, pumps and other equipment. Resulting costs for a medium size sewage works have been reported to easily reach US\$100,000/year.

Phosphorus recycling

Different work on the chemistry of struvite precipitation is reported, including a table of the **solubility constants** K_{sp} calculated by different authors. These vary significantly, probably as a result of pH and wastewater chemistry. Struvite is generally found to be least soluble (and so most likely to precipitate) at around pH9. Struvite solubility is particularly affected by the presence of calcium ions. The different models of struvite solubility (“Struvite”, “Minteq+”, and Mustovo *et al.* kinetic model) are discussed.

Struvite recovery from waste waters as a route for phosphorus recycling is presented. Struvite recovery is achieved by adding magnesium and/or increasing pH, the latter either by chemical addition or by aeration. Key experimental and pilot reactor experiences are referenced and tables show examples of waste streams with their struvite recovery potential and recovery costs as calculated by different authors.

Materials surfaces

The second paper “Struvite formation and the fouling propensity of different materials”, presents experimental results concerning struvite precipitation rates on different materials’ surfaces: stainless steel, acrylic (polymethyl methacrylate) and Teflon (polytetrafluoroethylene), smooth or roughened. The materials were attached to the stirrer in 1 litre beakers of anaerobically digested sludge centrate liquor from a sewage sludge treatment plant. The material surface was around 50 cm².

Stirring at 150 rpm proved sufficient to increase the pH of the liquors from pH 7 to pH 8.2. The rate of deposit of struvite on the stirrer material samples was measured over 8 hour runs, with samples being used for three consecutive experiments to assess how the deposit rate changed with the progressive formation of a layer of struvite. The acrylic and Teflon materials were tested both smooth as new, and roughened homogeneously with fine and with rough silicon carbide papers.

Struvite deposition was shown to be considerably lower on Teflon than on stainless steel, and

considerably lower again on acrylic (less than one tenth of the rate on steel). However, for all materials, deposition increased with reuse, showing that once initial struvite deposits have started to form this can accelerate the process. Roughening of the acrylic and Teflon materials considerably increased struvite deposition rates (bringing acrylic up to nearly half the rate for steel).

This confirms previous work showing lower struvite deposit rates on smoother materials in a study comparing iron, glass fibre reinforced plastic, techite and PVC.

The authors note that these results mean that although smooth materials may initially result in lower struvite deposit rates in sewage works this effect may well not be durable, as the material is roughened by sand and grit in sewage.

“Struvite formation, control and recovery”, *Water Research* 36, pages 3925-3940, 2002. J. Doyle¹, S. Parsons¹

“Struvite formation and the fouling propensity of different materials”, *Water Research* 36, pages 3971-3978, 2002. J. Doyle¹, K. Oldring², J. Chruchley², S. Parsons¹.

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Water Research:
<http://www.elsevier.com/locate/issn/00431354>

Water quality

Slovenia

Phosphorus load estimates

The authors used material flow analysis to develop a phosphorus balance for the lower Krka River basin in Slovenia. This basin has an area of 650 km² (3.2% of the land area of Slovenia) with a population of 52,000 inhabitants, including the strongly urbanised and industrialised Novo Mesto town area upstream. 46% of the land surface area is forest, and around one third cultivated agricultural land, the remainder being pastures and wetland.

During 1996-1997, 17 sampling surveys of water quality from 5 different main river locations were carried out. In parallel, material flow analysis was carried out to estimate the total phosphorus loads from wastewater treatment plants septic tanks and direct discharges of sewage into the river, agricultural and land run-off and atmospheric deposition.

Around one third of the catchment's domestic sewage was collected (17,000 households) and treated in an activated sludge sewage works (nutrient removal not installed).

For households with septic tanks, 95% of phosphates are commonly removed after passing through a few metres of soil (Gilliom and Clayton 1983).

A hypothesis of improved sewage treatment was developed, based on Slovenian law (1996) which requires collection of sewage and nutrient removal in sewage works (80% P-removal expected).

Agricultural phosphorus losses were estimated on the basis of 1.5 kg/hectare/year, based on typical figures given by the US EPA (1984). Livestock discharges were estimated at 1% of total annual manure phosphorus production. An improved land management scenario estimated that agricultural loadings could be reduced by 10% by Best Management Practices including soil conservation, nutrient management, feedlot management, removing most polluting fields from production, etc.

Half of P-loads from sewage

Total phosphorus loads to the Krka river were estimated to be 81.8 tonnesP/year, and this estimate was confirmed by the rivers sampling and flow data (see Drolc 1998). Wastewater discharges were estimated to make up just over half of phosphorus loadings (52.2%), mostly via sewage works discharges (49.5% of total loads) with leaching via septic tanks making up only 2.3% of the total P load.

Agriculture was estimated to contribute 44.6% of total phosphorus loads, mainly through diffuse sources (only 3.9% of total from point sources – direct discharge of manures). Atmospheric deposition was also estimated to contribute 3.2%.

Wastewater treatment

The hypothesis of upgrading of sewage treatment according to current Slovenian law led to an estimated reduction in wastewater phosphorus loads from 42.7 to 16.1 tonnesP/year, that is a 62%

reduction. If at the same time, reductions in agricultural phosphorus loads were made by Best Management Practices and changes in land-use, this would reduce agricultural loads by around 10%.

In this case, wastewater would then provide only one third (33.1%) of total phosphorus loads to the river, with an overall reduction in all loadings of 18% being achieved.

“Estimation of sources of total phosphorus in a river basin and assessment of alternatives for river pollution reduction”. Environment International 28 (2002) pages 393-400 www.elsevier.com/locate/envint

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Baltic Sea

Dissolved phosphorus linked to water movements

The Baltic Sea is semi-enclosed and brackish. Saltier, and so denser, water can flow into the Baltic from the North Atlantic, over the shallow sills of the Danish Straits, displacing the deep bottom water of the Baltic. 96 major inflows of this type have been recorded over the last century, but only a few since the mid-1970's and only two since 1980 (in 1983 and 1993).

The absence of such inflows is thought to have resulted in a “stagnation” of the bottom waters of the Baltic.

However, the widely used term “stagnation” appears to be inappropriate. The authors show that, contrary to expectations, the mixing of the Baltic was in fact greater during the long “stagnation” period 1983-1993. They suggest that the absence of major inflows reduces the salinity gradient in the Baltic, facilitating vertical mixing (the deep waters are not sufficiently more saline = heavier than upper waters to resist turbulent diffusion). Further, in between the major inflow events there have been many smaller inflows, which bring oxygenated water into the Baltic but may not enable the establishment of the salinity gradient which inhibits vertical mixing.

The authors show that during the prolonged “stagnation” 1983-1993, the area of hypoxic bottom waters (oxygen depleted) fell to only 5% of total

bottom water area (from a peak of 27% in the early 1970's). Also in this "stagnation" period, intermediate depth water (80-120m) had higher oxygen concentrations.

Implications for phosphorus

It is well documented that sediments can act as "sinks" for phosphorus from water when oxygen is available (in particular through P binding to iron) but that the phosphorus may be released back into the water in conditions of low or zero oxygen availability (hypoxic or anoxic conditions).

The authors show that, for the Baltic, changes in the pool of available phosphorus in the sea (water DIP = dissolved inorganic phosphorus pool) are effectively coherent with the changes in the area of hypoxic bottom water, and that oxygen concentrations were significantly negatively correlated to DIP concentrations in the hypoxic bottom waters. The DIP accumulation during expansion of the area of hypoxic bottom waters is about the same as the DIP losses during contraction.

Over the period of the study, total annual phosphorus loadings to the Baltic have varied within the range 23,000-37,000 tonnesP/year, without significant overall time trends. Although there have been reductions in fertiliser use and reductions in discharges through improved sewage treatment, these have not in fact substantially reduced total external phosphorus loadings (reductions of <1,000 tP/year).

Furthermore, the Baltic is estimated to export some 17,000 tP/year to adjacent sea basins, including 11,000 tP/year to the North Sea through the Danish Straits.

The variations in external P loads cannot thus explain the annual variations in the water phosphorus pool (DIP).

The authors conclude that large reversible variations in sediment phosphorus storage, related to climate events and water movements rather than to anthropogenic P loadings, will make it more difficult over the short term to manage phosphorus as the limiting nutrient in the Baltic.

"Hypoxia in the Baltic Sea and Basin-Scale Changes in Phosphorus Biochemistry", Environ. Sci. Technol. 2002, n°36, pages 5315-5320.

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Michigan, USA

Nitrogen addition increases stream periphyton growth

Six different nutrient addition concentrations were tested for impact on periphyton response (algae growing on stream-bed surfaces) in continuously flowing natural streamwater: three N:P molar ratios (65:1, 17:1, 4:1) at high and low nutrient concentrations. Each combination was subject to 5 simultaneous replicates (total 30).

The experiments were run for 28 days using the University of Michigan Biological Station Stream Research Facility. This offers 30 parallel once-through "flumes" (2.8m long, 10cm wide, 7cm deep vinyl gutters) through which water from East Branch Maple River is continuously pumped.

Natural stream water

This stream flows through forests, draining Douglas Lake and receiving considerable groundwater inflows, thus offering stable concentrations of nutrients. **The stream's nutrient concentrations, without N or P addition, corresponded to the high N:P ratio / low total nutrient concentration configuration** (dissolved inorganic nitrogen DIN around 43 µgDIN/l, soluble reactive phosphorus SRP around 1.4 µgSRP/l). The low and medium N:P / low total nutrient concentration (TNC) configurations were obtained by adding soluble phosphorus to the flume inflow (using peristaltic pumps), and the high TNC configurations by adding soluble N (up to around 140 µgDIN/l) as well as soluble P.

Water velocity in the plumes was maintained at 35-40 cm/s, temperature was around 12-17°C and artificial shading was used to reduce light intensity to around 15% of incident light comparable to the natural forest shading of the stream itself.

Periphyton increase with nitrogen addition

Development of periphyton was examined on 70 previously sterilised clay tiles (49 x 35 x 10 mm) placed in each plume. Periphyton was collected from

randomly chosen tiles at days 9, 17 and 28 and chlorophyll-*a*, total periphyton biovolume and species composition were assessed. The 28 day period was long enough for the complete cycle of periphyton appearance, growth and sloughing off to have taken place.

Higher TNC resulted in significant increases in periphyton chlorophyll-*a* and biovolume at day 9, but with this effect being diminished by day 28, because of loss of biomass through sloughing by this time. **The addition only of P (medium and low N:P ratios, low TNC) did not give any increase in periphyton biovolume or chlorophyll-*a* at day 9, and not clearly at day 17. Both biovolume and chlorophyll-*a* were, on the other hand, higher with high N additions.** The authors indicate that this was “contrary to our prediction that P would limit periphyton growth based on the high N:P ratio in the source ratio and unamended periphyton mats”.

The dominant algal group was diatoms (93% of periphyton biovolume). Of 11 common algal taxa counted, the relative abundance of 9 was affected by the changes in nutrient ratio or of total nutrient concentrations. Algal community structure was generally more sensitive to the changes than the overall algal volume.

The ratio N:P in the algal biomass mirrored closely changes in the water N:P ratio, driven mainly by changes in biomass %P. The %N in the algal biomass mirrored water N concentrations and was unaffected by the water N:P ratio.

The authors note that the stream N:P ratio (around 63:1) considerably exceeds the “Redfield Ratio” of 106C:16N:1P. The Redfield Ratio was empirically determined for oceanic seston, but is widely used for eutrophication modelling. Habitual eutrophication thinking would thus expect phosphorus to be the “limiting” nutrient in the stream. The authors conclude that, contrary to this expectation, algal growth was stimulated by N and not P concentrations: algal chlorophyll-*a* and biovolume were high in all high-N experiments and low in all low-N experiments, whereas increasing P concentrations within a given level of N concentration did not result in increased algal biomass.

“Effects of N:P ratio and total nutrient concentration on stream periphyton community structure, biomass and elemental composition”. Limnol. Oceanogr. 46(2) 2001 pages 356-367.

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Lowland river

Impact of sewage nutrient removal

A three-year study of the lowland river Wensum, Norfolk, England, published by the UK Environment Agency, examines dissolved and sediment phosphorus concentrations before and after sewage works P-removal implementation. Aquatic plant (macrophyte) populations were studied to assess the impact of this change in phosphorus loadings on populations.

The river Wensum is a prime example of a *Ranunculus* dominated calcareous lowland river specified in the EU Habitats Directive, one of England's 27 SSI (Site of Special Scientific Interest) designated rivers, and a proposed Special Conservation Area. The catchment studied (upper river) was 571 km².

Around 50 point sources of phosphorus were identified in the catchment, with a total phosphorus load of 85 kgTP/day from sewage works and 43 kgTP/day from industrial point sources (mainly food processing industries). Of these, **two municipal sewage works, Fakenham (19 kgTP/day) and East Dereham (32 kgTP/day), providing dominant contributions before implementation of P-removal.** After implementation, phosphorus discharges from these sewage works were respectively reduced to 6.95 kgTP/day (overall 77% P-removal) and 9.35 kgTP/day (88% P-removal).

Diffuse agricultural phosphorus loads to the Wensum river were estimated by using figures for per hectare releases from a dozen previous studies of similar catchments in Norfolk (figures quoted ranged from 0.075-0.101 to 0.231 kgTP/hectare/year). This corresponded to a predicted concentration of 53 µgTP/l in the river, exceptionally low for this area of England at 12% of total phosphorus loadings to the river, increasing to **32% of total-P from agriculture** after P-removal implementation in the two sewage works.

The Wensum also has four tributaries unaffected by known point sources of nutrients, providing a valuable comparison for the study of plant communities, and of impacts of diffuse nutrient loadings. Mean concentrations of total phosphorus in these sub-catchments were 39-77µgTP/l, presumably corresponding to diffuse loadings.

Impacts of phosphorus removal

The point sources of phosphorus along the river were found to considerably influence the in-water phosphorus concentration, whereas the **levels of phosphorus in sediments were much more influenced by the presence of weirs** (phosphorus levels in sediments much higher immediately upstream of weirs, in the pool of impounded water, than immediately downstream).

The P-removal at these two principal sewage works proved not sufficient to bring dissolved phosphate concentration levels down to target levels for such a river (EPV0 10-60 µg/l). However, **concentrations may be buffered by release of sediment phosphorus**, so that effects may become significant only after a few years. The levels of diffuse loads would also be an obstacle to achieving these objectives.

Macrophyte plant communities not correlated to nutrient concentrations

Plant communities were studied at 62 sites in the catchment, to compare populations in waters with different nutrient concentrations. **Aquatic vascular plants populations characteristics, including density, species composition, species traits and community structure, were not found to be correlated significantly with nutrient concentrations** over the range of soluble phosphorus 11-3600 µgSRP-P/l and ammonium 16-434 µgNH₄-N/l. Sediment phosphorus availability was also not significantly correlated with plant characteristics.

“Assessment of the Impact of Nutrient Removal on Eutrophic Rivers” Environment Agency R&D Technical Report P2-127/TR., November 2002. B. Demars, D. Harper, University of Lancaster.

<http://www.eareports.com/ea/rdreport.nsf/e92ef76ffa1cc0b680256975002e8822/5cf5e368dce2e6a380256cda0043207d?OpenDocument>

European Union Workshop

Defining eutrophication criteria

The EU Commission’s “Workshop on Eutrophication Criteria” (28-30 May 2002), chaired by the EU Directorate General for Environment, brought together scientific experts and representatives of the Member States. A clear and practical definition of “eutrophication”, for different types of surface waters, is necessary for the implementation of key EU water legislation, in particular the Urban Waste Water Treatment Directive (91/271), the Nitrates Directive (1991/676) and the new Water Framework Directive (2000/60). The Workshop’s objective was to start the process of establishing such an operational definition of “eutrophication”, in order to enable Member States to decide and to engage appropriate actions to protect and restore surface waters as required by these Directives.

The Urban Waste Water Treatment Directive (91/271) requires Member States to identify “*eutrophication sensitive areas*” and to implement nutrient removal from sewage for all conurbations of >10,000 person equivalents in these areas (and also in smaller conurbations where appropriate). This Directive includes a definition of eutrophication:

“The enrichment of waters by nutrients ... causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the water balance of organisms ... and to the quality of the water concerned”.

This definition thus clearly indicates that “**eutrophication**” **must be assessed from the endpoints of ecological and water quality changes**, and not simply on the basis of nutrient chemistry or even just of increased algal development (if this does not have detrimental impacts).

Definition and implementation

The Workshop agreed that this definition provided a good starting point. However, the EU Commission considers that “*Member States so far show some reluctance in designating sensitive areas*” and has recently drawn attention to the inadequacy of sensitive area designation in several states, including by taking certain states to the European Court (see

SCOPE Newsletters n°s 46, 47 – 2002 at www.ceep-phosphates.org).

This is indicative of the need to develop, starting from this outline definition of “eutrophication”, operational criteria applicable to different types of water body.

The **Nitrates Directive** (1991/676) further requires Member States to monitor the eutrophication status and eutrophication symptoms, effects and trends of “vulnerable areas”. This again requires an operational definition of “eutrophication” and of criteria and monitoring parameters for these criteria.

Eutrophication and “good ecological status”

The **Water Framework Directive** (2000/60) brings EU water legislation into a coherent integrated system, based on concerted catchment management and on the legal obligation to define and achieve “good ecological status” in all surface waters by 2015 (with the exception of heavily degraded/ modified water bodies), whilst maintaining the obligations of the Urban Waste Water and the Nitrates Directives. This therefore necessitates defining what “good ecological status” means in terms of eutrophication.

The challenge, as for the other aspects of water and biological quality, is to establish European guidelines for defining eutrophication which both allow harmonised European implementation of the Directive, monitoring and reporting, but also allow catchment managers to be responsive to local ecological specificities.

The Water Framework Directive’s definitions relevant to eutrophication are coherent with that of the Urban Waste Water Directive. “Good ecological quality” is defined for phytoplankton (example of “lakes”) as changes not “*resulting in undesirable disturbance to the balance of organisms ... or to the physico-chemical quality of the water or sediment*”, and for nutrients “*concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and ... the biological quality elements*”.

Taking these endpoint-based definitions as starting points, the Workshop concluded “It is important to use an operational approach when defining ‘eutrophication’, rather than a strict etymological approach. This means that **the attribute ‘eutrophicated’ should be given to ecosystems that exhibit deleterious effects of excessive primary**

production (intense phytoplankton and/or macrophyte proliferation, increased toxic phytoplankton events, hypoxia in the bottom waters), but **not to ecosystems where there is only nutrient enrichment**, but no present or potential noxious effects on the ecosystems and/or on water use.”

The Workshop thus agreed **an “operational definition” of eutrophication** could be: “**A new state of enrichment of the ecosystem, compared with pristine conditions, creating clear degradation or nuisances both for human uses of water and for general water quality (shift of species with loss of biodiversity, flora and fauna morbidity).**”

The Chairman therefore concluded: “*The remaining task would be to move from definitions to a common understanding of the acceptable level of shift of structure and function of the ecosystem, compared with reference conditions and the degree of acceptable adverse indirect effects on the water quality and water use.*”

Eutrophication criteria for different types of surface water

Participants at the workshop emphasised that: “*It is not only the specific nutrient concentrations as such that are important, but also the balance or ratio between Si:N:P and the deviation from the natural ratio. Natural eutrophication occurs (non-anthropogenic) all over the world; the difference between this and the man-made eutrophication is the speed at which they develop and, in many cases, the differences in magnitude of variation (amplitude).*”

The reactions to nutrient enrichment and the significance of related changes in phytoplankton and other ecosystem dynamics vary considerably between different types of surface waters. This is also true of many other water quality factors, and the EU is therefore currently developing a classification of surface water types with as an objective Water Framework Directive implementation guidelines for each type. For example, some 100 different types have been identified just for coastal waters, with the objective being to rationalise this down to 30-50 types. An European Commission Joint Research Centre project currently underway aims to map a “**eutrophication risk index**” for all of Europe’s coastal waters.

In some cases the **costs of achieving the “good ecological status”** required by the Water Framework

Directive may be unreasonably high, and it may thus be appropriate to try to assess what status of ecological quality can be economically achieved. This corresponds to the concept of “**Thresholds**”. See article in SCOPE Newsletter n° 48 at www.ceep-phosphates.org

Case studies

A number of case studies of eutrophication problems in European waters and attempts to address these were presented. These showed the **difficulty of defining nutrient levels for ecosystem restoration or management**, and the complexity of ecosystem reaction to nutrients and to other man-made changes.

In several cases of inland lakes (Lade Constance, Lake Geneva, Lac du Bourget France, Neusiedlersee Austria, Lakes Mjosa and Gjersjoen Norway ...) considerable reductions of phosphate inputs have been achieved over the last 15-20 years, essentially through sewage treatment (nutrient removal), but phosphate inputs are still higher than “natural” loadings. **Nitrogen inputs have not been so successfully reduced, because of loadings from agriculture.** Recovery in these lakes has been variable, showing the differences between different water bodies, and in many cases has been slower than hoped because of internal loadings and because of the practical difficulty in achieving nutrient loadings as low as or near natural, pristine conditions.

H. Fleckseder (Austrian Environment Ministry) presented the successful restoration of Lake Neusiedlersee, emphasising the importance of sewage nutrient removal: *“the total cost increment for that part of wastewater treatment that refers to the removal of nitrogen and phosphorus is relatively small compared with the cost of sewerage and wastewater treatment”*. The Lake’s restoration also included re-establishing balanced natural fish populations, buffer vegetation strips on agricultural land, and harvesting of reeds to remove organic production.

Speakers emphasised the variations of ecosystem response to changes in nutrient loadings, with other **factors such as climate and transparency having considerable effects on algal development, and zooplankton populations playing an important role (grazing of algal biomass)**. The Workshop indicated that zooplankton should be monitored in lakes (species composition and abundance) and questioned why zooplankton were not included in the

biological quality parameters for lakes listed in the Water Framework Directive.

For coastal/marine waters, OSPAR’s current approach of moving beyond the existing 50% nutrient reduction targets towards targeted eutrophication management of coastal ecoregions, also requires an operational definition of eutrophication (OSPAR Common Procedure for the Identification of the Eutrophication Status of coastal and marine waters).

Case studies of coastal/ marine waters included the North Sea, the Baltic and the Mediterranean. Again, the considerable complexity of reaction to nutrient enrichment was apparent, with in coastal waters nitrogen often being the key limiting nutrient, but both nitrogen and phosphate levels and ratios having implications for development of toxic algal blooms.

The Workshop concluded that:

“It would be advisable not to propose absolute threshold values for nutrient concentrations, as they cannot be considered alone as reliable risk evaluators because the hydrodynamics and physical characteristics (depth, turbidity) modulate strongly the nutrient role.”

Eutrophication monitoring parameters

The Workshop considered that eutrophication monitoring should aim to assess whether the ecosystem is undergoing an “acceptable deviation” from reference ecosystem parameter levels. Eutrophication monitoring should therefore include:

- a) causal and basic response parameters:
 - * total nitrogen
 - * total phosphorus
 - * chlorophyll_a
 - * water transparency (Secchi disk depths)
- b) ecosystem response parameters:
 - * dissolved oxygen
 - * macrophytes
 - * phytoplankton index
 - (% blue greens, bloom frequency)
 - * fish index
 - * pH
 - * benthic community index
 - * autotroph index
 - * algal growth potential (bioassays)

“Workshop on Eutrophication Criteria”, Brussels 28-30 May, 2002. European Commission, Directorate General Environment (ENV.B1 – Water, Marine and Soil Environment)

Conference announcement

Eutrophication of waters from diffuse sources.

The 7th International Conference on Diffuse Pollution and Basin Management (*DiPCon*) 17-22nd August 2003, Dublin

There is increasing awareness that diffuse sources of nutrients are a major component of the eutrophication of surface waters, and that build up of both phosphorus and nitrogen in ground waters originate from activities throughout a catchment.

In recent years, widespread improvement in phosphorus removal facilities at waste water treatment plants and reduction in supply of phosphorus from detergents has necessitated a **reassessment of source of nutrients to freshwaters**. There is an increasing focus on the relationship between land management and nutrient supply from a variety of diffuse sources. These are both difficult to quantify and control.

Since the first conference in Chicago in 1993, the International Water Association has organised a series of conferences on diffuse pollution in *Prague (1995)*, *Edinburgh (1998)*, *Bangkok (2000)*, *Milwaukee (2001)* and *Amsterdam (2002)*. Parallel initiatives have occurred at national and international level. For example since July 1997 an EU COST (Co-operation in Science and Technology) Action has been operative to (i) develop a common technical understanding of the processes, forms and pathways of P loss in land run-off, and (ii) develop a methodological framework for identifying the relative contribution of different diffuse P sources at the farm and catchment level (see <http://www.cost832.alterra.nl>). Recently the 3rd International Workshop on Phosphorus Transport took place in Plymouth, England (reported in *SCOPE Newsletter n° 46*).

This attention to diffuse sources of nutrient enrichment has begun to unravel some of the mysteries of phosphorus and nitrogen movement from catchment to water. We now know, for example, that the majority of phosphorus can come off the land in only a few storm events in the year; that the mechanism of nutrient movements can be quite different between arable and grassland systems; and that inherent properties of the soil, past-history of

fertilisation and antecedent moisture conditions can all be important controls on phosphorus transport. An equally complex, but different, set of factors controls net movement of nitrogen from land to water.

The new **Water Framework Directive (2000/60/EC)** requires that of activities within catchments that pose a risk to the maintenance of good water status are verified by monitoring, a programme of measures needs to be initiated that reduces impact on surface, ground and coastal waters. The implication for current agriculture and other activities, such as forestry, that provide the pressures to aquatic systems could be enormous.

Recent actions taken or initiated by the EU against member states for failure to comply with the Directives on Nitrates (*91/676/EEC*) and Dangerous Substances (*76/464/EEC*) suggest that compliance with the Water Framework Directive will also be carefully monitored.

The need, therefore, to understand the mechanisms that drive diffuse nutrient enrichment appears self-evident.

The 7th International Conference on Diffuse Pollution and Basin Management (*DiPCon*) 17-22nd August 2003, Dublin, will bring together researchers, practitioners and regulators from many countries, to share experience in understanding issues of diffuse pollution as it affects aquatic systems.

The conference in Dublin in 2003 will be the first time the regulatory aspects of pollution control will be given prominence. It will also be the first international diffuse pollution conference to have a major theme focusing on marine and transitional waters, in parallel with the traditional themes concerning urban and rural land-uses. 2003 also sees the start of the next major cycle of **EU Common Agricultural Policy (CAP) review**. The Dublin 2003 conference will provide an international forum to consider practical experience of agricultural support schemes with contrasting examples from North America, EU countries and elsewhere. The conference aims to play a role in developing best practice for World Trade Organisation compliant agricultural support mechanisms, consistent with state of the art understanding and management of environmental consequences.

The implications of the Water Framework Directive, which reflects international moves towards integrated environmental management, will provide a current

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management context to the conference and one which is of relevance to all parts of the world.

The *Dublin 2003 Conference* will focus on opportunities and solutions relating to the diffuse pollution control aspects of water management.

Conference themes include:

- Integrated Water Resource Management (including impact of EU Water Framework Directive)
- Policy and Socio-Economic Considerations
- Agricultural Pollution Abatement Strategies (e.g. Cross Compliance)
- Sustainable Urban Drainage Systems (SUDS)
- Forestry and Mining Issues
- Maritime and Coastal Area Issues
- Groundwater and Baseflow Protection
- Aquatic Ecology and Dangerous Substances
- Organisation for Basin Management (including Transboundary Pollution Management)
- GIS Modelling and Remote Sensing
- Public Consultation and Awareness/Education
- Regulatory Framework

The 7th International DiPCon Conference on
<http://www.ucd.ie/~dipcon/dipcon.htm>



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IWA Specialist Group
Management –
International Conference
Norwegian University
of Science and Technology (NTNU)

**BIOSOLIDS 2003 Wastewater Sludge as a
Resource**

Trondheim, Norway 23-25 June, 2003

Programme includes:

sludge minimisation, sludge characteristics, sludge dewatering, pre-treatment and aerobic digestion, treatment technologies for land use – hygiene, case studies, **phosphate recovery (Wed. 25th June, 8h30-10h25)**, energy recovery, recovery of specific products, final handling of biosolids and future perspectives.

Phosphate recovery session includes: Recovery of phosphate from sludge – a Dutch analysis, Phosphorous and coagulant recovery by alkaline sludge treatment, Phosphorus recovery by one- or two-step technology with use of acids and bases, Recycling of sludge with the Aqua Reci – process

<http://www.ivt.ntnu.no/ivm/Biosolids2003/>

The SCOPE Newsletter

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The SCOPE Newsletter seeks to promote the sustainable use of phosphates through recovery and recycling and a better understanding of the role of phosphates in the environment.

The SCOPE Newsletter is open to input from its readers and we welcome all comments or information. Contributions from readers are invited on all subjects concerning phosphates, detergents, sewage treatment and the environment. You are invited to submit scientific papers for review.

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