# STRIVE

Report Series No.91

# Water Quality and the Aquatic Environment

## STRIVE

Environmental Protection Agency Programme

2007-2013

Environmental Protection Agency
An Ghníomhaireacht um Chaomhnú Comhshaoil

SSTI Strategy for Science, Technology and Innovation

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

# Environmental Protection Agency

The Environmental Protection Agency (EPA) is a statutory body responsible for protecting the environment in Ireland. We regulate and police activities that might otherwise cause pollution. We ensure there is solid information on environmental trends so that necessary actions are taken. Our priorities are protecting the Irish environment and ensuring that development is sustainable.

The EPA is an independent public body established in July 1993 under the Environmental Protection Agency Act, 1992. Its sponsor in Government is the Department of the Environment, Community and Local Government.

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We license the following to ensure that their emissions do not endanger human health or harm the environment:

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- Office of Climate, Licensing and Resource Use
- Office of Environmental Enforcement
- Office of Environmental Assessment
- Office of Communications and Corporate Services

The EPA is assisted by an Advisory Committee of twelve members who meet several times a year to discuss issues of concern and offer advice to the Board.

**EPA STRIVE Programme 2007–2013**

# Water Quality and the Aquatic Environment

An Assessment of Aquatic Ecosystem Responses to Measures Aimed at Improving Water Quality in the Irish Ecoregion

## (2007-W-MS-3-S1)

## STRIVE Synthesis Report

***End of Project Report available for download on http://erc.epa.ie/safer/reports***


Prepared for the Environmental Protection Agency

by

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## ACKNOWLEDGEMENTS

This report is published as part of the Science, Technology, Research and Innovation for the Environment (STRIVE) Programme 2007–2013. The programme is financed by the Irish Government under the National Development Plan 2007–2013. It is administered on behalf of the Department of the Environment, Community and Local Government by the Environmental Protection Agency which has the statutory function of co-ordinating and promoting environmental research.

A successful outcome to EFFECT would not have been possible without the support and help of numerous individuals and institutions. We would like to thank in particular Alice Wemaere of the EPA and external members of the project steering committee for their guidance and support throughout the project and all those who assisted with field and laboratory work and the production of figures. Thanks are also due to several individuals and institutions, notably the Shannon Regional Fisheries Board (Inland Fisheries Ireland), for permitting use of their data and to the numerous landowners and estate managers who facilitated access to field study sites.





## EPA STRIVE PROGRAMME 2007–2013

Published by the Environmental Protection Agency, Ireland

ISBN: 978-1-84095-445-6

Price: Free **Online version**

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# Executive Summary

Anthropogenic eutrophication of rivers and lakes has become a major and persistent problem throughout the Irish Ecoregion. In response, measures aimed at reversing eutrophication and its effects have been implemented. These measures largely target the mitigation of inputs of phosphorus (P) and other nutrients. However, few studies have been carried out into the suitablity and effectiveness of these measures and the factors that potentially influence recovery of rivers and lakes following their implementation. Focusing on the Irish Ecoregion, the EFFECT[1] project aimed both to better understand the role of environmental conditions in mediating the effectiveness of measures aimed at reducing P and other nutrient inputs, and to determine the effects on surface water quality (rivers, streams and lakes) of their implementation in different geographic settings.

Work Package (WP) 1 determined relationships between environmental conditions and P concentrations. Lough Sheelin in the Republic of Ireland (RoI) was selected for detailed investigation. Data for the period 1995–2008 indicated declining P concentrations, with the extent of poorly drained soils, cattle-stocking densities and runoff having the strongest positive influence on concentrations. Malfunctioning septic tank systems in the catchment may also have acted as point sources of P, thereby diminishing the effectiveness of measures aimed at diffuse agricultural sources. Despite a trend of reduced external loadings overall, P concentrations in the lake remained at early 1990s and higher levels through to 2008, possibly because of the activities of invasive zebra mussels (*Dreissena polymorpha* Pallas) and internal loading from sedimentary sources. Work Package 1 also investigated the relationships between environment and P conditions at a smaller geographic scale. Two databases were constructed, both comprising flow-weighted P concentrations for river monitoring sites for the period 2006–2008: one comprised sites from the RoI only; the other also included stations in Northern Ireland (NI). The strongest predictors of concentrations of P in rivers in the databases were human population density, extent of artificial surfaces, runoff risk, percentage of pasture density of livestock (cattle) (all positive), and mean catchment slope, drainage density extent of forestry (all negative). Geology (in particular the susceptibility of bedrock to weathering) also influenced P concentrations.

Work Package 2 examined the biological and chemical impacts of measures aimed at reducing P losses from agriculture implemented in NI since the 1990s. Water quality measurements determined in the 1990s for 42 low-lying streams in two catchments (Colebrooke and Upper Bann) in NI were used as a baseline and compared with data from 2008–2009. Monitoring in the 1990s revealed that both biological and chemical indices of water quality were strongly correlated. By 1998 chemical water quality (CWQ) had improved significantly, although no improvement in biological water quality (BWQ) was evident. A resurvey in 2008–2009 revealed a continued improvement in CWQ, but no widespread, consistent improvement in BWQ. Agricultural land use intensity was a significant predictor of BWQ: of particular note is a possible threshold where livestocking rates above and below one dairy cow $ha^{-1}$ are associated with, respectively, poor and good BWQ. Hydromorphological modifications, notably channelisation, the invasive amphipod *Gammarus pulex* (L.), intermittent pollution from farms and poorly functioning septic tank systems associated with recent housing developments in rural areas were also identified as possible constraints on recovery.

Work Package 3 investigated the association between riparian measures and water quality and ecological functioning in upland streams in afforested peatland catchments in NI. Catchment-scale characteristics, including land use and average slope, had a greater influence on water quality than the structure and composition of riparian vegetation. River plants and invertebrates appeared to be influenced by different environmental factors. Invertebrates were affected by pH and ammonium, whereas macrophtyes were more strongly related to altitude, nitrate and the amount of

1 The EFFECT (An Effective Framework For assessing aquatic ECosysTem responses to implementation of the Phosphorous Regulations) project was funded by the EPA Ireland, 2008–2011.

peatland vegetation near the streams. Compared with afforested sites with no management measures, streams with some form of riparian buffer exhibited higher macroinvertebrate biomass and diversity.

Work Package 4 was located in the Blackwater catchment, which straddles the NI–RoI border. A voluntary scheme aimed at mitigating P impacts at low flows had been introduced in the study area in the late 2000s. The scheme involved replacing the most defective septic tank systems associated with rural housing. Work Package 4 evaluated the effectiveness of the scheme. An initial survey was carried out prior to the replacements being made in 2007, and again following the installation of the new septic tank systems. Some positive effects on water quality were evident where there was little change in the density of systems. However, in parts of the catchment where there were increases in the total number of septic tank systems overall during the survey period, no significant improvements in water quality were evident. This finding is of particular concern given that other efforts to mitigate P inputs were also implemented during the study period.

Mixed results for the project overall indicate a need for improved understanding of the processes influencing the production and transport of pollutants and their impacts on aquatic ecosystems. Narrow strips of riparian vegetation were found to be ineffective buffers against the effects of extensive changes in catchment conditions, such as those associated with conifer plantations. Moreover, some rivers and lakes have been so profoundly modified hydromorphologically and by the presence of invasive taxa that physical remodification may be required, while discharges from septic tank systems have been identified in three of the four EFFECT WPs as likely constraints on the effectiveness of measures aimed at reducing P inputs to water bodies. Future research in the Irish Ecoregion should target catchments characterised by relatively well-drained soils. The latter were under-represented in the current project, which largely focused on impermeable soils where pollution problems are often at their most acute. Finally, attention should be paid in future work to the degree that POMs have been and are being implemented.

# 1 Introduction to the EFFECT Research Project

Widespread deterioration in water quality as a result of anthropogenic activity has led to the development and implementation of legislation at national and international levels. Recently, national legislation of European Union (EU) member states has been subsumed within the Water Framework Directive (WFD) (2000/60/EC), which seeks to ensure the effective and sustainable management of water resources, and to achieve and maintain good water quality for all water bodies by 2015 (Anon, 2005). The WFD management unit, River Basin District (RBD), encourages an integrated catchment-scale approach to water quality management (Rekolainen *et al.,* 2003; Bennion and Battarbee, 2007). Management plans drawn up by RBDs have to include the design and implementation of programmes of measures (POMs) that are needed to ensure that the water quality objectives of the WFD are met within the stipulated timeframe. Programmes of measures devised to mitigate pollution impacts, including the impacts of nutrients (Crabtree *et al.*, 2009), are largely based on existing European regulations and policies.

Anthropogenic eutrophication, resulting from over-enrichment by nutrients and in particular phosphorus (P), is a major cause of deteriorating water quality (Smith and Schindler, 2009). Although point sources of P (for example, waste water treatment plants [WWTPs]) are important, diffuse sources of P from agriculture have been identified as the main cause of nutrient enrichment in freshwaters (Jennings *et al.*, 2003; Sharpley *et al.*, 2009), and continue to prove a significant challenge to water-quality improvement efforts in the Irish Ecoregion (mainly comprising the island of Ireland). Since the 1980s and 1990s the proportion of water bodies classed as having moderate quality has increased (McGarrigle *et al.,* 2010), in part due to a decline from good status: by the mid-2000s about 90% of water bodies in Northern Ireland (NI) were thought to be at risk of not making the WFD objectives (EHS, 2005), while in the Republic of Ireland (RoI) 64% of rivers and 38% of lakes were thought 'at risk' and 'probably at risk' (Anon, 2005).

EU Directive (91/676/EEC), commonly known as the Nitrates Directive (Howarth, 2006), plays a key role in the Irish Ecoregion in the delivery of water-quality improvements required by the WFD. In the RoI, the WFD followed publication of a strategy document setting out the Government's approach to reducing P inputs (DoE, 1997). As part of the strategy, the Local Government (Water Pollution) Act, 1977 (Water Quality Standards for Phosphorus) Regulations, 1998, S.I. No. 258/1998, more commonly known as the Phosphorus Regulations, or P Regs, were introduced. As with the WFD, the P Regs require that water quality be maintained or improved according to biological and chemical quality indices and with reference to baseline conditions. On 1 January, 2007, new legislation came into operation aimed at improving the management of nutrients, including P, on farms in NI, updating legislation and schemes introduced in the 1990s. The new legislation (the Nitrates Action Programme Regulations [Northern Ireland] 2006 and the Phosphorus [Use in Agriculture] Regulations [Northern Ireland] 2006) applies to all farmers in NI (Table 1.1).

To date, reviews of P Regs-related POMs have focused on levels of implementation and compliance. Separately, periodic reviews of water quality in surface and groundwaters have been released in both RoI and NI. Relatively little attention has been paid to water quality changes following the implementation of particular POMs, however, or to improved understanding of those factors upon which a successful implementation may be contingent. The EFFECT[1] project sought to address this knowledge gap by evaluating changes in chemical water quality (CWQ) and biological water quality (BWQ) of a sample of rivers and lakes in the Irish Ecoregion (Fig. 1.1) following implementation of a range of POMs aimed at reducing P and other nutrient inputs to water bodies. Pollutants such as P have multiple sources

1 The EFFECT (An Effective Framework For assessing aquatic ECosysTem responses to implementation of the Phosphorous Regulations) project was funded by the EPA Ireland, 2008–2011.

**Table 1.1. Summary of key legislation and measures relating to agriculture for Northern Ireland, 1989–2009.**

| Date | Action | Detail |
|---|---|---|
| 1989 | Capital Grants available to farmers (DARD) | Construction and upgrade of waste-handling facilities on farms to address point source pollution |
| 1989–1992 | Shift towards wilted silage due to changes in farm machinery-reduced effluent volumes | Non-statutory silage pre-wilting began |
| 1991 | Control of Pollution (Silage, Slurry and Agricultural Fuel Oil) Act | Act was not statutory in Northern Ireland until 1994. Stipulated standards for silage and slurry storage |
| 1997–1998 | Erne Nutrient Management Scheme | Provided targeted nutrient management advice including soil and grass testing |
| 1999 | Responsible Phosphate Management Scheme | Aimed to increase farmer awareness of the phosphate problem in Northern Ireland and increase corrective actions |
| 2003 | EC Water Framework Directive (WFD) | WFD becomes part of NI legislation |
| 2004 | The Common Agricultural Policy Single Payment and Support Schemes (Cross Compliance) Regulations(NI) 2005 | Maintain land according to Good Agricultural and Environmental Condition requirements |
| | | Respect statutory management requirements (e.g. EC Directives) to receive monetary support |
| 2004 | NI Farm Quality Assurance Scheme | Environmental Cross Compliance |
| | | Follows Codes of Good Agricultural Practice |
| | | Uptake ~11,000 beef and sheep farmers |
| 1991 | EC Nitrates Directive | EC Nitrates Directive |
| 2003 | EC Nitrates Directive | Codes of Good Agricultural Practice |
| 2004 | The Protection of Water Against Agricultural Nitrate Pollution Regulations (NI) 2004. | Total Territory approach taken in Northern Ireland: Entire region classed as Nitrate Vulnerable Zone |
| 2007 | Nitrates Action Programme (NI) 2006 | Phased implementation of Nitrates Directive Action Plan from 2007. Full operation of closed season for spreading of livestock wastes from January 2009 |
| 2007 | Phosphorus Regulations (NI) 2006 | Required that use of chemical P fertilisers is in accordance with crop need and takes into account the availability of manure P |
| 2007–2008 | Farm Nutrient Management Scheme (DARD) | Grant aid. Construction of manure and slurry storage facilities to enable farms to meet closed season slurry storage requirements. Total cost > £200m |

within the landscape, while the processes governing their mobilisation vary spatially (e.g. as a result of differences in soil properties and land use) and temporally (e.g. due to seasonal variations in rainfall). EFFECT sought to capture this variability while examining a range of different POMs in a variety of environmental settings.

The following is a brief overview of the EFFECT project, the major findings and the most important implications for policy-makers. A much more thorough description of the project, including the individual work packages (WP) and related data and analyses, is available in the full, final report (Taylor *et al.,* 2012).

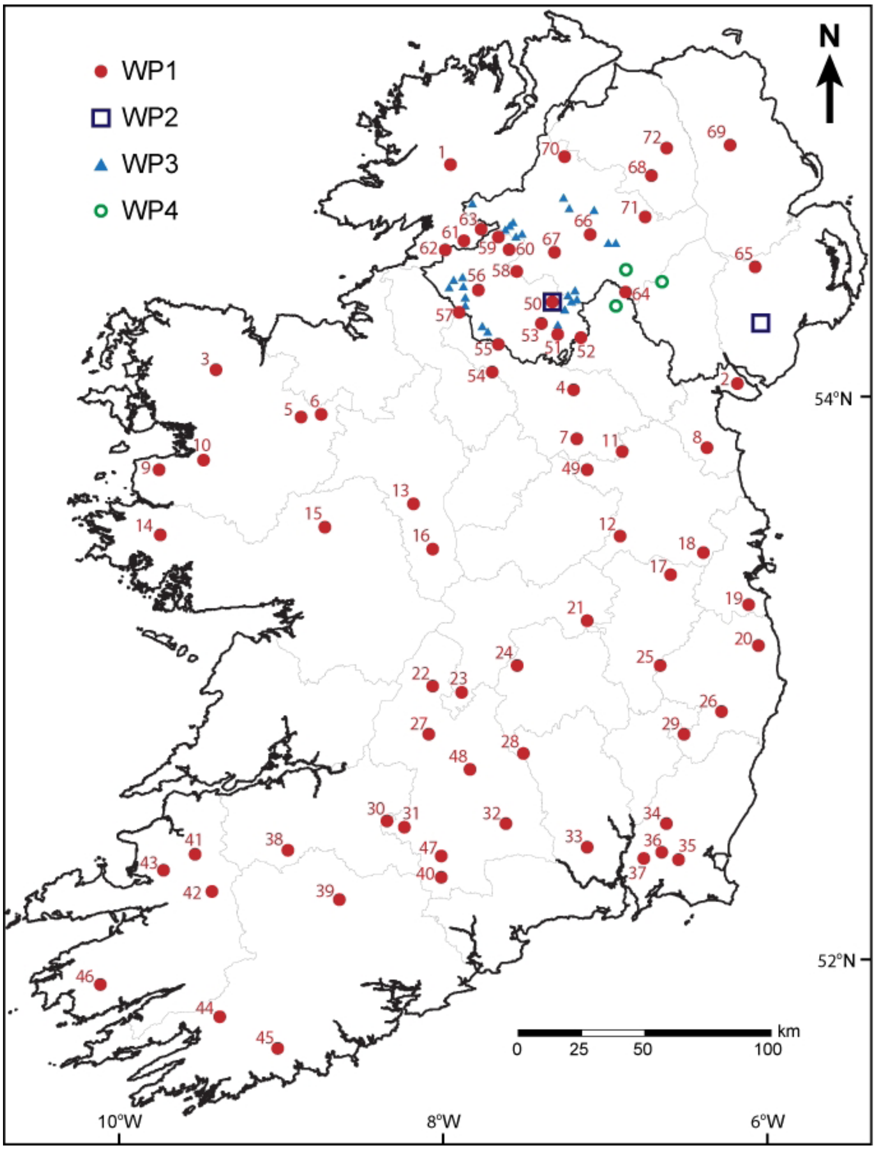


**Figure 1.1. Map of the Irish Ecoregion showing the location of the study sites upon which Work Packages (WP) 1–4 were based. Numbers for WP 1 sites correspond with Table 2.1 a, b (and see Taylor *et al.*, 2012).**

# 2 Summary of Approaches, Methods and Main Results by Work Package

## 2.1 Work Package 1: Environmental Influences over Phosphorus Concentrations in Rivers and Lakes in the Irish Ecoregion and Implications for Effectiveness of Phosphorus-related Programmes of Measures

The null hypothesis ($H_o$) that there is no relationship between environmental conditions and P concentrations in rivers and lakes was tested at both large (subcatchment) and small (catchments in RoI and in RoI and NI combined [the Irish Ecoregion]) scales. The large-scale/small-area study focused on the catchment of Lough Sheelin in the RoI and also permitted analysis of the efficacy of POMs targeting P in rivers and Lough Sheelin. Zebra mussel (*Dreissena polymorpha Pallas)* populations were established in Lough Sheelin by 2004 (Kerins *et al.*, 2007), with initial colonisation presumably commencing after implementation in 1998–1999 of a P Regs programme aimed at mitigating levels of P entering water bodies in the catchment. In an attempt to distinguish the actual impacts of implementing P Regs, monitoring data from before the onset of zebra mussel colonisation (1990–1999) were used as a control reference against which to compare P dynamics post zebra mussel establishment (2004–2008). The period 1990–1999 includes an earlier attempt of P mitigation (1990–1992) in the catchment that largely involved restrictions on the spreading of manure.

The small-scale/large-area study utilised river water quality and environmental data from 72 catchments in the RoI and NI.

### *2.1.1 Material and Methods*

The large-scale/small-area component of WP1 used total phosphorus (TP) and molybdate reactive phosphorus (MRP) concentration data from seven river monitoring stations in the Lough Sheelin catchment. A load apportionment model (LAM) and TP budget were also constructed for Lough Sheelin and its catchment and provided a means of identifying key factors influencing P loadings.

The small-scale/large-area component of WP1 used river water quality and river catchment attribute data from 49 catchments in the RoI and 23 in NI (Tables 2.1a and b). Simple linear regression was applied to a RoI database and to a combined RoI-NI database to determine the strength of relationships between flow-weighted MRP data (fwMRP) as a dependent variable and individual catchment attributes as independent variables. Principal components analysis (PCA) was applied to the predictor variables to yield standardised, independent predictors of fwMRP. Two sets of three principal components were estimated for each of the two databases. Optimum combinations of predictors were selected and four independent regression models calibrated. The coefficient of determination ($R^2$) quantified the proportion of variation explained by the calibrated models ($p < 0.05$).

Geospatial models were developed from a combination of the linear regression models of fwMRP and environmental predictor variables with corresponding variograms. A kriging function was then applied, and the predicted variance of estimates and accuracy of predictions for the geospatial models calculated. The geospatial models were validated using data from five river catchments in the RoI not included in the original analysis.

### *2.1.2 Results*

Data for the seven Lough Sheelin subcatchments included in the study for 1995–2008 indicate a trend of declining exports of MRP – independent of any long-term, sustained changes in runoff. A distinct seasonal cycle of P concentration was evident in the dataset, with highest P concentrations in the summer to autumn months, corresponding with lowest river discharge (Fig. 2.1). Results from the LAM indicated input of P from both point and diffuse sources, although substantial inter-annual and inter-subcatchment variations in TP and MRP export rates were evident.

**Table 2.1a. Locational information for catchments in Republic of Ireland (RoI) used in the small-scale/large-area study. Catchment numbers correspond to those used on Figure 1.1. Data from EPA, RoI.**

| Catchment number | River name | EPA river monitoring point | EPA reference code | Catchment area ($km^2$) | Easting | Northing |
|---|---|---|---|---|---|---|
| 1 | Finn (Donegal) | Bridge 2.5 km u/s Ballybofey | 01F01-0600 | 309.74 | 212475 | 395036 |
| 2 | Big (Louth) | Ballygoly Bridge | 06B01-0100 | 10.56 | 315156 | 309883 |
| 3 | Deel (Crossmolina) | Knockadangan Bridge | 34D01-0300 | 226.77 | 115748 | 319214 |
| 4 | Erne | Kilconny Belturbet (RHS) | 36E01-1400 | 1491.19 | 236117 | 317097 |
| 5 | Swinford | Swinford: Bridge on Foxford Road | 34S05-0200 | 17.73 | 136632 | 300450 |
| 6 | Charlestown stream | Bridge northwest of Bellahy | 34C28-0100 | 25.20 | 147505 | 302542 |
| 7 | Mountnugent | Mountnugent Bridge | 26M02-0500 | 91.03 | 248916 | 285680 |
| 8 | White (Louth) | Coneyburrow Bridge | 06W01-0500 | 55.23 | 305719 | 289280 |
| 9 | Bunowen (Louisburgh) | Bridge in Louisburgh | 32B03-0150 | 70.37 | 80678 | 280689 |
| 10 | Carrowbeg (Westport) | Cooloughra Bridge | 32C05-0100 | 36.04 | 102283 | 282745 |
| 11 | Blackwater (Kells) | Liscartan Waterworks | 07B01-1600 | 698.54 | 284248 | 269593 |
| 12 | Boyne | Ballinter Bridge | 07B04-1600 | 1575.64 | 289519 | 262675 |
| 13 | Hind | Bridge east of Ballymartin | 26H01-0300 | 44.11 | 188040 | 261794 |
| 14 | Recess | Canal Bridge | 31R01-0500 | 111.82 | 80248 | 247482 |
| 15 | Clare (Galway) | Claregalway Bridge | 30C01-1200 | 1072.39 | 137185 | 233239 |
| 16 | Cross (Roscommon) | Bridge S of Doyle’s Bridge | 26C10-0300 | 103.27 | 201201 | 240231 |
| 17 | Lyreen | Just u/s Rye Water | 09L02-0100 | 87.62 | 294316 | 238709 |
| 18 | Tolka | Violet Hill Drive Finglas | 09T01-1100 | 133.21 | 314313 | 237425 |
| 19 | Shanganagh | At Commons Road | 10S01-0600 | 32.52 | 325296 | 222967 |
| 20 | Newtownmountkennedy | Bridge south of Ballyphilip | 10N02-0600 | 16.19 | 329594 | 206255 |
| 21 | Barrow | Pass Bridge | 14B01-1000 | 1063.12 | 262245 | 210990 |
| 22 | Ballyfinboy | Bridge just u/s Lough Derg | 25B02-0800 | 186.46 | 183804 | 198048 |
| 23 | Little Brosna | Milltown Bridge (near Mount Lucas) | 25L02-0100 | 113.84 | 206992 | 190924 |
| 24 | Delour | Derrynaseera Bridge | 15D01-0400 | 69.68 | 229478 | 192448 |
| 25 | Greese | Bridge near Greese Bank | 14G04-0200 | 49.41 | 279834 | 195747 |
| 26 | Aughrim (Wicklow) | Coat’s Bridge | 10A02-0200 | 188.18 | 314795 | 178944 |
| 27 | Nenagh | Bridge near Tyrone Abbey | 25N01-0500 | 136.29 | 187567 | 178055 |
| 28 | Goul | Fertagh Bridge | 15G02-0300 | 118.10 | 230637 | 170320 |
| 29 | Slaney | Scarawalsh Bridge | 12S02-2200 | 1030.71 | 298379 | 145074 |
| 30 | Mahore | Bridge at Riverville | 24M04-0900 | 44.28 | 168953 | 137988 |
| 31 | Ara | Railway Bridge (Tipperary town) | 16A03-0300 | 44.18 | 189787 | 135151 |
| 32 | Anner | Bridge near Anner House | 16A02-1100 | 444.70 | 224468 | 123193 |
| 33 | Blackwater (Kilmacow) | Dangan Bridge | 16B02-0300 | 109.98 | 257030 | 119868 |
| 34 | Boro | Ballynapierce Bridge | 12B02-0500 | 173.90 | 295600 | 136400 |
| 35 | Mulmontry | Goff’s Bridge | 13M01-0700 | 56.64 | 287163 | 118531 |
| 36 | Corock | Bridge east of Foulkesmills | 13C01-0100 | 62.57 | 285387 | 118696 |
| 37 | Owenduff | Taylorstown Bridge | 13O01-0240 | 103.09 | 282075 | 114627 |
| 38 | Deel (Newcastlewest) | 0.8 km u/s Castlemahon Bridge | 24D02-0500 | 261.89 | 131806 | 130995 |
| 39 | Blackwater (Munster) | Ballyduff Bridge | 18B02-2500 | 2333.87 | 196540 | 99120 |
| 40 | Tar | Ford u/s Tar Bridge | 16T01-0600 | 229.65 | 210816 | 113399 |
| 41 | Smerlagh | Ford u/s Feale R confl (LHS) | 23S02-0700 | 127.94 | 102540 | 132324 |

Continued overleaf

**Table 2.1a. Locational information for catchments in Republic of Ireland (RoI) used in the small-scale/large-area study. Catchment numbers correspond to those used on Figure 1.1. Data from EPA, RoI. (*continued*)**

| Catchment number | River name | EPA river monitoring point | EPA reference code | Catchment area (km$^2$) | Easting | Northing |
|---|---|---|---|---|---|---|
| 42 | Shanowen (Maine) | Ford Bridge u/s Maine River confluence | 22S01-0100 | 41.11 | 101349 | 109064 |
| 43 | Big (Tralee) | At Dunnes Stores | 23B04-0150 | 10.65 | 84050 | 114850 |
| 44 | Owvane (Cork) | Pierson's Bridge (LHS) | 21O07-0400 | 72.13 | 102397 | 54522 |
| 45 | Argideen | Jones Bridge | 20A02-0100 | 78.71 | 140467 | 44445 |
| 46 | Cummeragh | Dromkeare Bridge | 21C04-0600 | 47.99 | 54537 | 68551 |
| 47 | Thonoge | Bridge u/s Ballylooby | 16T02-0080 | 19.15 | 200424 | 119604 |
| 48 | Suir | Bridge near Suirville House (New Bridge) | 16S02-1600 | 1090.80 | 200215 | 134186 |
| 49 | Inny | Ballinrink Bridge | 26I01-0300 | 58.30 | 249453 | 280948 |

u/s = upstream. RHS = right-hand side, LHS = left-hand side

**Table 2.1b. Locational information for catchments in Northern Ireland (NI) used in the small scale/large area study. Catchment numbers correspond to those used on Figure 1.1. Data from the NI Environment Agency (NIEA).**

| Catchment number | River name | NIEA river monitoring point | NIEA reference code | Catchment area (km$^2$) | Easting | Northing |
|---|---|---|---|---|---|---|
| 50 | Colebrooke | Ballindarragh Bridge | 10715 | 314.14 | 233100 | 336000 |
| 51 | Newtownbutler | Newtownbutler | 10675 | 10.45 | 241800 | 325900 |
| 52 | Finn (Tyrone) | Wattle Bridge | 10728 | 274.16 | 242500 | 320300 |
| 53 | Hollybrooke | Aghalurcher | 10714 | 29.30 | 236300 | 331100 |
| 54 | Woodford | Aghalane | 10734 | 396.65 | 234200 | 319400 |
| 55 | Swanlinbar | Thompsons Bridge | 10735 | 104.42 | 225300 | 331300 |
| 56 | Sillees | Thompsons Bridge | 10747 | 151.82 | 218100 | 344800 |
| 57 | Arney | Drumane Bridge | 10737 | 247.91 | 217500 | 337500 |
| 58 | Ballinamallard | Ballycassidy Bridge | 10700 | 166.73 | 222800 | 350700 |
| 59 | Bannagh | Bannagh Bridge | 10681 | 33.88 | 216200 | 365400 |
| 60 | Kesh | Kesh Bridge | 10688 | 88.53 | 218000 | 363900 |
| 61 | Waterfoot | Letter Bridge | 10665 | 36.47 | 208500 | 365200 |
| 62 | Garvary | Larkhill | 10663 | 16.23 | 200900 | 363000 |
| 63 | Termon | Tullyhommon | 10679 | 61.98 | 211000 | 366700 |
| 64 | Blackwater (Benburb) | Benburb | 10330 | 963.98 | 281900 | 352000 |
| 65 | Lagan | Shaws Bridge | 10512 | 492.64 | 332500 | 369000 |
| 66 | Camowen | Donnellys Bridge | 10111 | 276.17 | 246400 | 373000 |
| 67 | Drumragh | Campsie Bridge | 10128 | 318.38 | 245800 | 372400 |
| 68 | Moyola | Moyola new Bridge | 10380 | 305.84 | 295600 | 390500 |
| 69 | Main (Dunmore) | Dunmore Bridge | 10212 | 706.97 | 308700 | 389600 |
| 70 | Burndennet | Burndennet Bridge | 10022 | 146.32 | 237400 | 404800 |
| 71 | Ballinderry | Ballinderry Bridge | 10361 | 433.24 | 292700 | 379800 |
| 72 | Clady | Glenone Bridge | 10438 | 125.23 | 296300 | 403800 |

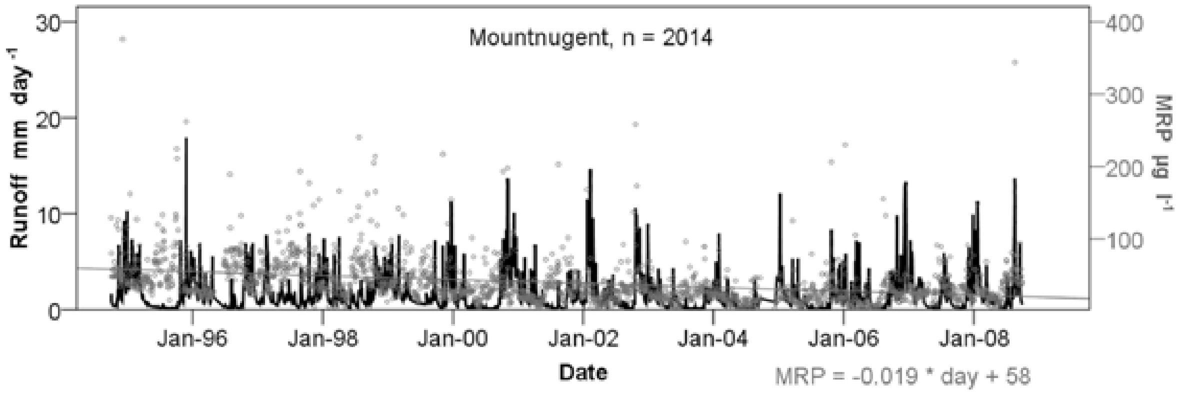


**Figure 2.1. Example plot (Mountnugent subcatchment) from large-scale/small-area component of Work Package 1 showing seasonal and annual variations in measured runoff (mm day$^{-1}$) and molybdate reactive phosphorous (MRP) (µg l$^{-1}$) concentrations (1995–2008) with sample size (n). The solid black lines represent measured runoff rates and the grey hollow circles are measured MRP concentrations. The simple linear equation for empirical MRP trend is shown under the x axis.**

Soil cover was found to have an important influence on P loads; subcatchments with poorly drained gleys as the predominant soil cover were associated with the highest overall accumulated P loads from diffuse sources. MRP and TP from diffuse sources were also positively correlated with cattle-stocking densities and to levels of human population. The latter most likely reflects septic tank soak-aways acting as diffuse sources of P.

Time series data were divided into Stage 1 (1995–1998, pre-implementation of P mitigation measures) and Stage 2 (1999–2008, post-implementation of P mitigation measures). Generally, annual loads of P from diffuse sources exceeded those from point loads. Reductions of TP and MRP from point sources in one of the subcatchments (Mountnugent) in 1999–2000 likely reflect increased removal of P following improvements in the treatment of waste water. However, extreme runoff events contributed a large proportion of annual P load to rivers in all of the study subcatchments. For example, in 2007–2008, days with runoff greater than 10 mm day$^{-1}$ accounted for 35% of the annual P load in Mountnugent, and reversed an established year-on-year trend of reducing contributions of P from diffuse sources. Although overall rainfall levels were similar compared with previous years, 2007–2008 was characterised by an anomalously high frequency of storms in which large amounts of precipitation fell in a relatively short period of time.

Annual flow-weighted TP concentrations ($TP_{in}$) entering Lough Sheelin and TP concentrations in the lake ($TP_{lake}$) for 1990–2008 demonstrate the effects of restrictions on the spreading of livestock manure during winter months, introduced in the early 1990s. The relationship between $TP_{in}$ and $TP_{lake}$ shows a marked change from the late 1990s. Substantial decreases in $TP_{in}$ following implementation of P Regs in the catchment are not reflected in $TP_{lake}$ concentrations. Annual averages of $TP_{lake}$ followed a similar trend to annual averages of chlorophyll *a* and secchi depth for the period 1990 to 1999. However, the chlorophyll *a*-TP relationship altered between 1999 and 2004 (Fig. 2.2); chlorophyll *a* concentrations declined post-2004 and therefore did not correspond with reduced TP concentrations following establishment of zebra mussels.

The TP budget for Lough Sheelin (1990–2008) comprised inflow TP ($TP_{load}$), change of TP storage in the water column, outflow TP ($TP_{out}$) and net TP retained/released by the lake sediments ($TP_{net}$). Sediments in the lake functioned as a net sink for TP in all years except 2004, when net TP release occurred. Total P loading showed substantial annual fluctuations. The most accurate models of TP loading for the lake for the period 1990–1999 were used as a baseline against which to compare changes in $TP_{lake}$ following establishment of zebra mussels by 2004. A change in direction of trends in both predicted and measured $TP_{lake}$ data was evident, with levels of both declining post 2004.

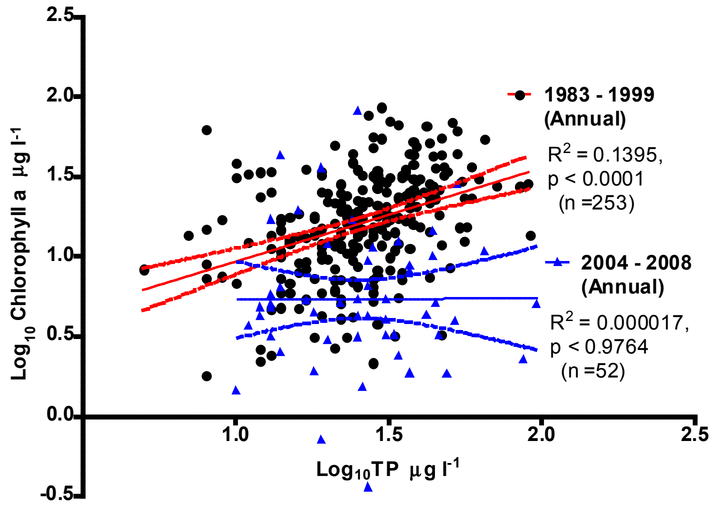


**Figure 2.2. Relationship between measured $\log_{10}$ chlorophyll a and $\log_{10}$ total phosphorus (TP) concentrations for 1983–2008 based on annual data for Lough Sheelin (large-scale/small-area part of Work Package 1). The red regression line shows the long-term relationship between $\log_{10}$ chla and $\log_{10}$TP before the invasion of zebra mussels (1983–1999) and the blue line shows the relationship between the two parameters after the establishment of zebra mussels (2004–2008).**

For the small-scale/large-area component of WP1, including both NI and ROI, simple linear regression identified a number of significant environmental predictors of fwMRP (Table 2.2). The same predictors were also prominent in the principal components of variation, as identified through PCA. Slight differences in results between the two databases (RoI and combined RoI-NI) were evident. The dominant predictors of fwMRP concentrations in study catchments in the RoI were catchment mean slope, the degree of soil saturation and level of human population. However, for the combined RoI and NI database, catchment mean slope was found to be a poor indicator of MRP and level of human population a good predictor. Overall, the variables % pasture, cattle density, % urban, runoff risk, % artificial, human density, % bedrock susceptible to weathering and degree of soil saturation were positively related to fwMRP. Furthermore, % bedrock resistant to weathering, catchment mean slope, drainage density and % forest were inversely related to fwMRP. Drainage density was significantly related to fwMRP in the RoI database, but not in the combined RoI–NI database.

Both the winter and summer geospatial (RK) models for the two databases produced higher $R^2$ values than their corresponding regression model. The winter geospatial model based on the RoI database generated the lowest root mean square error (RMSE) overall. Results of the validation exercise revealed that the winter model based on the RoI database has a much higher prediction accuracy than the corresponding summer model, with the latter appearing to overestimate transfers of MRP to rivers when compared with observed levels.

Adding a measure of spatial dependence as a predictor of fwMRP had a greater influence on the strengths of the winter models than the summer models. This finding may reflect a greater spatial correlation between fwMRP concentrations at river monitoring points during the hydrologically more variable months, influenced by ecological and hydrological processes acting at different spatial and temporal scales.

Taken together, the results provide a sound basis for rejecting the $H_0$ that underpinned WP1: environmental factors clearly have a strong influence over P concentrations in water bodies in the Irish Ecoregion. The identification of environmental predictors of P concentration in rivers enabled the development of geospatial models that have potential for use in identifying rivers with a high likelihood of being vulnerable to impairment by P or relatively resistant to recovery following reduced inputs of P.

**Table 2.2. Significant predictors of seasonal flow-weighted concentration of MRP (μg $l^{-1}$) in the Irish Ecoregion.**

| Predictor (catchment attribute) | Description | Database | Data type |
|---|---|---|---|
| ArtificalLX1 | CLC class - % of artificial surfaces in catchment (log x+1) | Both databases | Continuous |
| Bed[1,2,3]LX1 | % of bedrock type in catchment | Bed1LX1(combined RoI-NI database), Bed3LX1 (RoI database) | Continuous |
| Cattle | Cattle density in catchment (cattle $km^{-2}$) | Both databases | Continuous |
| DrainageDensityLX1 | Length of rivers (km) $km^{-2}$ | RoI database only | Continuous |
| ForestLX1 | % extent forestry in catchment | Both databases | Continuous |
| HumansLX1 | Human population in catchment (person $km^{-2}$) | Both databases | Continuous |
| MeanSlopeLX1 | Mean slope of catchment (%) | Both databases | Continuous |
| Pasture | CLC class – % of pasture in catchment | Both databases | Continuous |
| RunoffRiskLX1 | P runoff risk index applied to % extent of gley soils in catchment | Both databases | Continuous |
| TWILX1 | Topographic Wetness Index (dimensionless) | RoI database only | Continuous |
| UrbanLX1 | % of urban area in catchment | Both databases | Continuous |

CLC = corine landcover, P = phosphorus

## 2.2 Work Package 2: The Long-term Effectiveness of Programmes of Measures at the Farm Level

Work Package 2 tested the $H_0$ that the implementation of measures aimed at reducing agricultural pollution of water bodies in NI has had no effect on CWQ or BWQ. Specifically, WP2 employed CWQ and BWQ survey data collected in the 1990s as a baseline for assessing the impacts of farm level, nutrient management-based POMs on observed water quality in 2008–2009. The work focused on two catchments in NI: the Colebrooke and the Upper Bann.

The entire Colebrooke catchment was designated by the UK government as a Less Favoured Area (LFA) owing to the poor agricultural quality of soils overall. Land use is predominantly beef cattle, but includes large tracts of unimproved uplands, peatlands and coniferous forestry at higher altitudes. The Upper Bann catchment drains part of the western Mourne Mountains. Land use in the lowlands of the catchment is largely intensive dairy farming on mineral soils, with sheep farming more common in the upland parts of the catchment, where wetter, more organic soils predominate. Half of the catchment has LFA designation.

### *2.2.1 Materials and Methods*

Work Package 2 required a re-sampling of the monitoring network in the Upper Bann and Colebrooke catchments that was operational during the 1990s. Chemical water quality parameters and nutrients were sampled at fortnightly intervals over the 2009 hydrologic year. P and nitrogen (N) were analysed using standard methods (Gibson *et al.*, 1980; Jordan, 1997) with soluble fractions determined by 0.45 μm filtration. The Biological Monitoring Working Party (BMWP) system was employed for BWQ sampling (Chesters, 1980; BMWP, 1981). Nutrient concentrations were combined with flow data to derive annual flow weighted mean concentrations (fwMCs) to reduce the influence of variations in rainfall. The study was linked to geographic information system (GIS)-based assessments of land-use and land-use intensity based on the farm census returns of the Department of Agriculture and Rural

Development (DARD) and the CORINE land cover classification. In order to account for differences in runoff volumes between years, the study compared fwMCs of nutrients, rather than nutrient loads.

### *2.2.2 Results*

Work Package 2 demonstrated that mitigation measures implemented in NI since the early 1990s, and largely before the implementation in 2007 of the Nitrates Action Programme (NAP), have significantly reduced agricultural nutrient losses of N and P (Fig. 2.3), with concurrent improvements to CWQ. While part of the improvement was due to reduced incidences of agricultural point source pollution, the extent to which a Farm Nutrient Management Scheme, implemented in 2007–2008 under the NAP, will yield further improvements in respect to controlling diffuse pollution remains to be seen.

There were also less encouraging findings: in the Upper Bann exports of P increased between the late 1990s and 2009. In the absence of significant changes to agricultural intensity and hydrology, the expansion of rural housing and septic tank installations within this catchment over the last decade is implicated in the increase. However, marked reductions in P use by the dairy sector, via low P fertiliser use and a lower P content of livestock feeds, may have had insufficient time to be manifest in water quality assessments.

This study observed strong relationships between livestocking rates and BWQ, demonstrating the significant role of agriculture in determining the ecological status of these streams (Fig. 2.4). Critically, the scale of the overall improvement in BWQ was modest to non-existent compared with the improvement in CWQ. Nevertheless, the strength of these relationships has increased and they now predict greater BWQ at comparable livestocking rates compared with the 1990s. Of particular note is the potential existence of a threshold of agricultural intensity, where livestocking rates above and below 1 dairy cow equivalent (16.6 kg P $ha^{-1}$ $yr^{-1}$) are associated with poor and good BWQ, respectively.

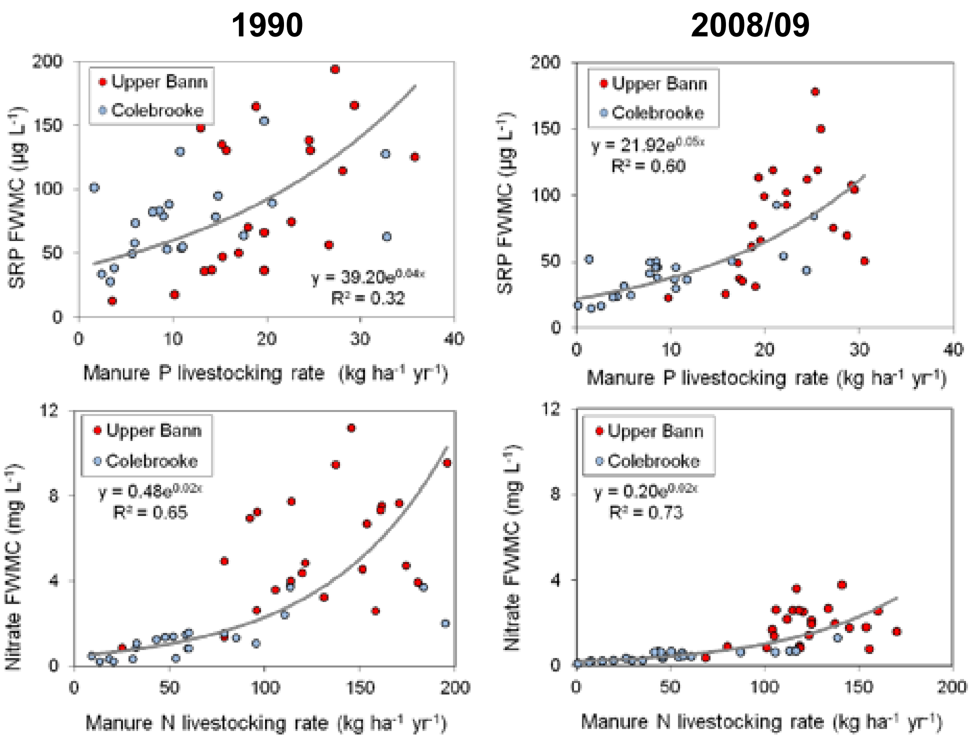


**Figure 2.3. Relationships between catchment livestock manure N and P stocking rate and annual flow weighted mean concentrations (FWMCs) of nitrate and soluble reactive phosphorus in the Colebrooke and Upper Bann catchments in 1990 and 2008/09 (Work Package 2).**

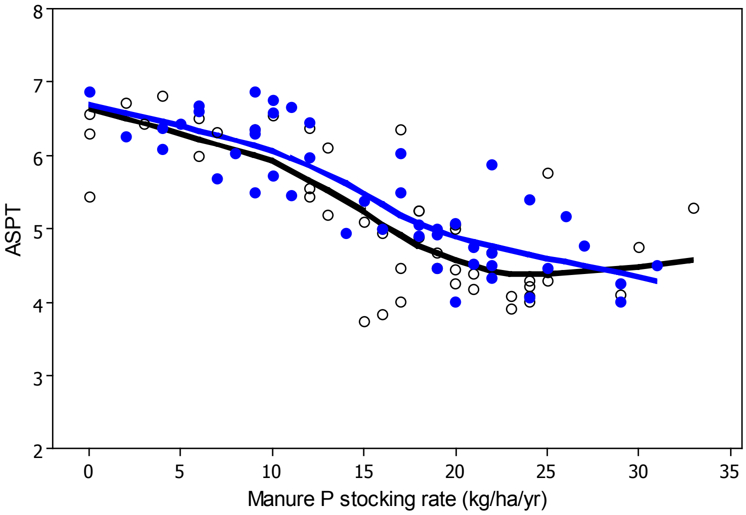


**Figure 2.4. Relationship between manure P livestocking rate and biological water quality as assessed by the Average Score Per Taxon (ASPT) (Work Package 2). Filled-in data points are for 2008/09 and unfilled data points are for 1998. Fitted lines are Lowess-smoothed curves.**

Improved CWQ over the period of study without a corresponding significant improvement in BWQ means that in this case the $H_0$ can only be partially rejected. Moreover, while the negative relationships between BWQ and stream P concentrations/livestocking rates indicate that agriculture is a strong driver, the processes by which it may impact BWQ are less obvious. Potential constraints to BWQ recovery, most of which can be correlated with agricultural land use intensity, are:

1. ***Hydromorphological constraints***: Stream hydromorphology appears better suited to lower-quality lentic biota: many tributaries, particularly in the Upper Bann, are channelised, and areas of riffles with which higher quality biotic elements are associated are rare.

2. ***Intermittent point source pollution***: Continuing and sporadic agricultural point source pollution may still constrain recovery in some streams, and such pollution may be more prevalent at higher livestocking densities. In addition, the potential for impairment by point source discharges from septic tanks exists – particularly during low summer flows.

3. ***Organic enrichment***: Organic enrichment favours the development of extensive biofilms, the oxygen demand of which is not assessed by standard analyses. Such biofilms are abundant in many of the most nutrient-rich streams, and these may have a high dark-oxygen demand, particularly during low summer flows. Furthermore, such biofilms may physically exclude higher-quality taxa by eliminating suitable habitats.

4. ***Seasonality***: Impacts of all of the above are known to vary according to season, largely because of seasonal differences in flow and temperature. Warmer temperatures and reduced precipitation predicted under various climate change scenarios are therefore likely to affect biotic recovery in the future.

5. ***Invasive species***: For example, the invasive crustacean amphipod *Gammarus pulex* (L.) generally dominates the invertebrate biomass at enriched sites and was recorded in the Upper Bann. Large numbers of *G. pulex* may prevent the recolonisation of higher-quality taxa. Organic matter enrichment of the benthos (point 3 above), which is not routinely assessed, may promote *G. pulex* abundance, thereby posing an additional constraint to the recovery of BWQ.

## 2.3 Work Package 3: The Impact of Programmes of Measures on Stream Water Quality, with Respect to Areas of Coniferous Forest

The planting of conifer forest close to the edge of headwater streams can lead to enhanced acidification (Evans *et al.,* in press) and an impoverished aquatic biota (e.g. O'Driscoll *et al.*, 2006). These effects may be mitigated through the management of riparian vegetation (Dosskey *et al.,* 2010). Work Package 3 assessed the impact of installing buffer strips of vegetation of different composition and structure on the ecological status of headwater streams, thereby testing the $H_0$ that riparian measures had no effects on CWQ and BWQ in headwater streams draining afforested peatland catchments.

Twenty-five first or second-order streams were examined in upland counties Fermanagh and Tyrone, NI. Streams were classified into five riparian management categories (RMC) according to the vegetation present on stream banks: *No buffer, Open buffer* and *Broadleaved buffer* (all with unharvested conifer plantations in the catchment with, respectively, conifers planted to stream edge, buffers of grasses, shrubs and occasional trees, and buffers comprising deciduous trees); *Harvested* (sites without buffers in catchments where harvesting of conifers occurred 12–24 months before sampling); and *Control* (catchments with undisturbed peatland vegetation and no conifer plantations).

### 2.3.1 Materials and Methods

Work Package 3 investigated differences between streams in control and forested catchments. The influence of vegetation – along stream banks and in the catchment – on water chemistry and stream ecology was examined and environmental correlates with benthic macroinvertebrate and macrophyte taxa composition identified. Streams were sampled over 50 m-long reaches. Food web structure was investigated using both carbon (C) and N stable isotope (SI) analysis and the functional feeding group (FFG) approach. Catchment vegetation and soil type were determined using GIS procedures with vegetation type and composition confirmed during subsequent field visits.

Water chemistry parameters were determined on three occasions for each stream (summer, autumn, winter). Benthic macroinvertebrates were sampled using a Surber sampler at 10 m intervals along each stream reach (at last six replicates): biofilm on rocks on the stream bed was also sampled at each 10 m station using a 10 cm quadrat. Kick sampling along the entire reach was undertaken after Surber sampling to obtain representative samples of invertebrates for SI analysis. Benthic organic matter was quantified using a Surber sampler. Macrophytes in the stream channel and bryophytes in the stream splash zone were recorded. A river habitat survey (after Raven *et al.,* 1998) was carried out at each stream site. Flow was recorded every 5 m and an average value was calculated for the stream. The mean reduction in light intensity (PAR) at the stream water surface compared with open conditions was calculated from measurements made at 5 m intervals along each stream reach sampled.

### 2.3.2 Results

Riparian measures had a relatively weak correlation with stream biota. Much stronger correlations were found between catchment-scale characteristics and chemical and biological stream water conditions. The results therefore do not provide a firm basis for rejecting the $H_0$.

Ammonium ($NH_4$) and pH explained most variation in the macroinvertebrate taxa composition of WP3 streams (Fig. 2.5). Decomposition of organic matter can result in high concentrations of $NH_4$ in some soils, where nitrification is inhibited by acidic or anaerobic conditions (Tamm *et al.*, 1974; Pajuste and Frey, 2003), as can occur in peatland soils. In this study, $NH_4$ was correlated with harvesting in the catchment, catchment slope and suspended solids. The acidity of stream water is a recognised driver of invertebrate ecology. Riipinen *et al.* (2010) suggested that stream pH is more important than riparian vegetation in determining litter breakdown rates, a key stream function, and the occurrence of invertebrate shredder species. In this study, pH was found to be highly correlated with altitude and the extent of rough grazing. Although chemical predictors were correlated with factors such as harvesting, significant differences in pH or $NH_4$ were not apparent between the RMCs. Results suggest that the type of riparian buffer is correlated only weakly with invertebrate species composition within a stream. Land use at the catchment scale was more strongly correlated with water chemistry than local riparian scale factors.

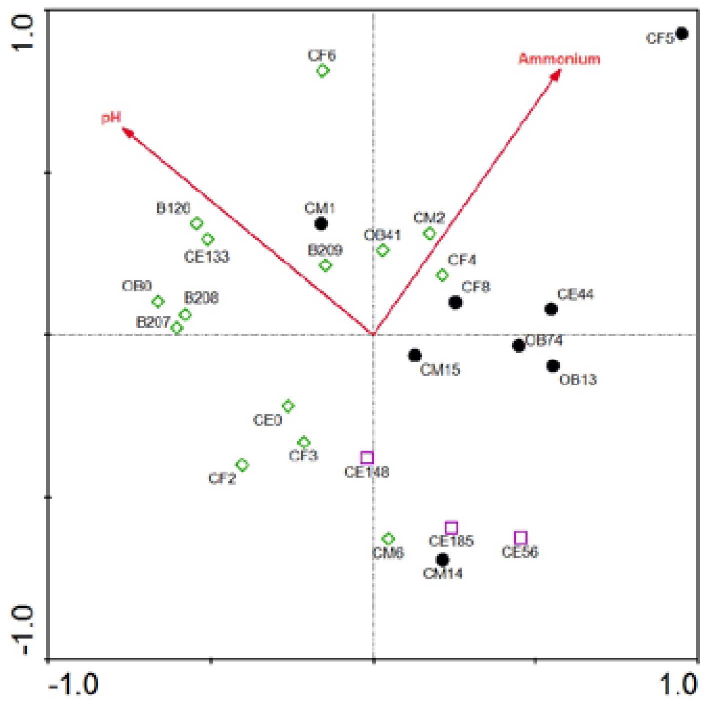


**Figure 2.5. Redundancy analysis of invertebrate biomass in 25 streams in Northern Ireland showing the two most influential variables as identified by multidiscriminant analysis (Work Package 3). The abbreviations CE, CM, B, OB and CF refer to No buffer, Control, Broadleaved buffer, Open buffer and Harvested, respectively. Different symbols represent *a posteriori* biological classification of streams.**

Altitude, nitrate ($NO_3$) and peatland vegetation in the riparian zone were found to have the strongest correlation with macrophyte species occurrence and abundance (Fig. 2.6). The majority of macrophyte taxa were negatively correlated with peatland vegetation in the riparian zone. This may be explained by the unstable splash zone habitat at peatland-dominated reaches. Stable sites, especially in relation to substratum, favour bryophytes (the dominant macrophyte group in all streams studied in WP3), as they take a long time to establish (Steinman and Boston, 1993; Suren, 1996). In concurrence with Mykra *et al.* (2008), the community structure of macrophytes and invertebrates appears to respond to different stressors.

Nutrient status, while only determined from three spot samples (for summer, autumn and winter), was influenced more by catchment slope than other catchment or riparian variables, with catchment slope negatively correlated with nutrient concentrations. A lower angle of catchment slope indicates longer groundwater residence times, facilitating the development of larger areas of saturated soil leading to increased dissolved nutrient concentrations in the water; water moving rapidly down steep slopes inhibits dissolution of soil organic matter (Rasmussen *et al.*, 1989; D'Arcy and Carignan, 1997; Prepas *et al.*, 2006).

According to the SI approach used in WP3, the invertebrate biomass at most sites was predominantly derived from terrestrial matter (allochthonous), but less abundant species obtained a high proportion of their C from autochthonous (in-stream) sources. These latter species generally comprised the greatest proportion of the total invertebrate biomass at the control and harvested sites. Streams with broadleaved and open buffers tended to contain the highest proportions of species predominantly using terrestrially derived matter. Results of SI analysis also showed that light reduction by riparian vegetation, catchment slope and the mass of organic biofilm were most strongly correlated with resource use by invertebrates (Fig. 2.7). Sample sites with high frequencies of allochthonous consumers tended to occur where the angle of catchment slope was relatively low (higher nutrient levels), shading was greater and organic biofilm more abundant. Stable isotope data also indicated that the number of niches for aquatic invertebrates was highest at sites where conifers had been harvested and where there had been no planting of riparian vegetation.

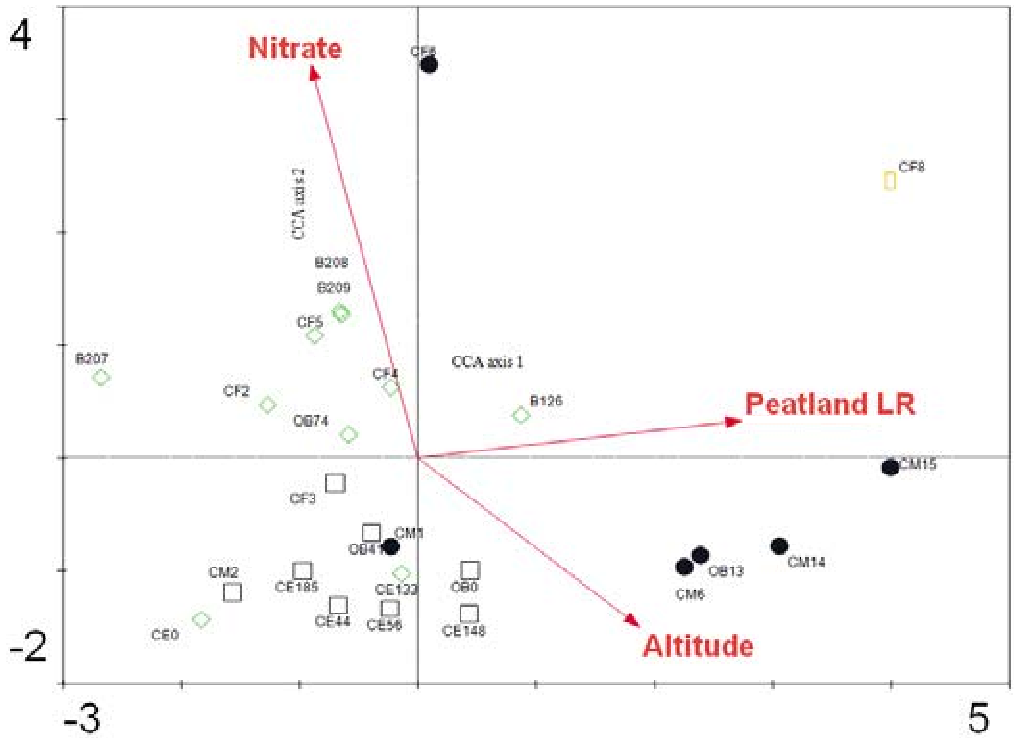


Figure 2.6. Canonical correspondence analysis of macrophyte taxa in 25 streams in Northern Ireland showing the three most influential variables as identified by multidiscriminant analysis (Work Package 3). Peatland LR – Percentage of peatland vegetation in the local riparian zone. The abbreviations CE, CM, B, OB and CF refer to No Buffer (conifer to edge), Control, Broadleaved buffer, Open buffer and Harvested sites respectively. Different symbols represent *a posteriori* biological classification of streams.

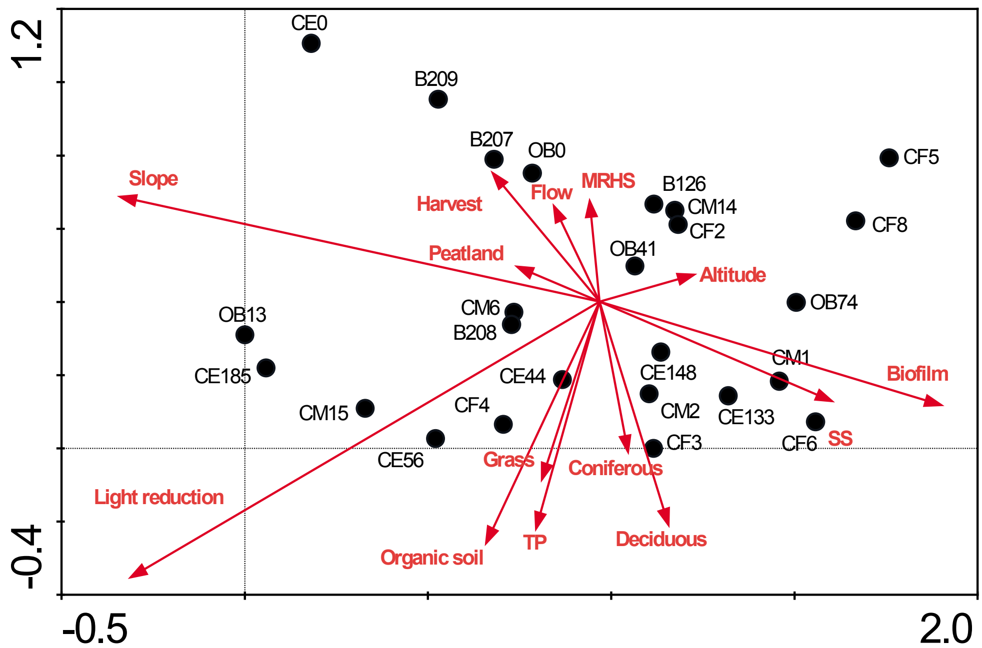


Figure 2.7. Detrended correspondence analysis for the proportion of consumers at each site within categories of percentage dietary reliance on terrestrially derived matter (allochthonous) where environmental variables are passively projected into the ordination space (Work Package 3). MRHS – river habitat survey, SS – suspended solids, TP – total P, slope – catchment slope. Land use characteristics are in percentage of catchment area.

Overall, the FFG and SI approaches yielded similar results with respect to the extent to which stream invertebrate biomass was sustained by allochthonous and autochthonous energy resources. The agreement stems from the fact that for the consumers dominating the biomass, resource utilisation according to SI and FFG agreed closely. Indeed, the SI approach was particularly useful for determining resource use among less abundant taxa, for which resource use often differs considerably from that indicated by mouth part morphology (or FFG), and therefore for identifying additional invertebrate niches. The SI approach therefore identified significant plasticity with respect to resource use, suggesting that generalism may be more prevalent among aquatic consumers. Of particular note in this respect is that at many sites, methane-derived C was indicated in the diets of several consumers, which was consistent with anaerobic conditions in adjacent C-rich riparian and hyporheic zones.

No significant difference was found between RMCs for benthic organic matter or algal metrics; any differences appear stream dependent. Moreover, invertebrate community structure and resource utilisation appear largely independent of riparian vegetation (Fig. 2.8). However, harvesting of conifers resulted in increased abundances of some taxa and greater energy mobilisation from in-stream sources.

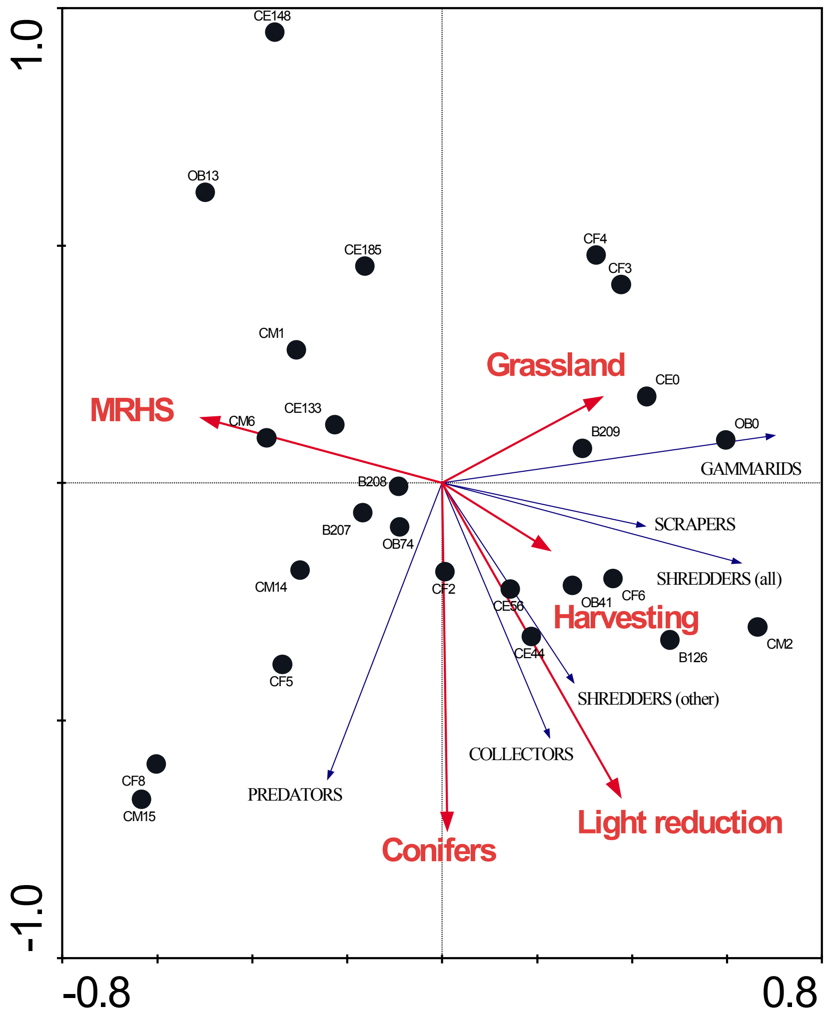


**Figure 2.8. Redundancy analysis for invertebrate biomass by functional feeding group for each study site, with the most influential variables, as indicated by multidiscriminant analysis, superimposed (Work Package 3). The abbreviations CE, CM, B, OB and CF refer to No buffer, Control, Broadleaved buffer, Open buffer and Harvested sites, respectively. MHRS – river habitat survey, Conifers, Harvesting, Grassland – percentage of catchment. Light reduction – percentage of light reduction.**

## 2.4 Work Package 4: Point Source (Septic Tank System) Mitigation in a Rural Catchment

Focusing on the Blackwater catchment, WP 4 tested the $H_0$ that a voluntary scheme to replace the most defective septic tank systems with state-of-the-art equipment, introduced between July and November 2007, had no impact on CWQ during low flows.

The Blackwater is the largest of six rivers flowing into hypereutrophic Lough Neagh, NI (Foy *et al.,* 2003). The study area comprised three subcatchments with similar drumlinised landscapes blanketed by gleyed soils (Cruickshank, 1997) and isolated from the aquifer by a layer of glacial till (Ó Dochartaigh, 2003). Land use is typically grassland for dairy and beef cattle and a relatively small number of sheep and several poultry units. The catchment also has a dispersed rural population, which is typical for this part of the Irish Ecoregion. The combination of gleyed soils, high drainage density (augmented by drainage schemes from fields to channels) and low evapotranspiration ensures efficient hydrological connection from land to water courses. Runoff is typically flashy with suppressed baseflows during summer (Wilcock, 1997). With similarities in land use and landscape, runoff is broadly comparable across the tributaries and headwaters of the catchment (Jordan *et al.,* 2005a).

### *2.4.1 Materials and Methods*

Stations consisting of rated water level recorders and bankside nutrient analysers monitored discharge and CWQ on a subhourly basis at the outlet of each subcatchment (Jordan *et al.,* 2005b, 2007; Cassidy and Jordan, 2011). Subsequent to the analysis described in Arnscheidt *et al.* (2007), several septic tank systems with a high risk of leaking pollution were chosen for replacement in each of the subcatchments studied. To account for interannual hydroclimatic variations influencing P transfer patterns, two full hydrological years (1 April 2006 to 31 March 2008) of P data were collected for comparisons with data from two years subsequent to replacement (1 April 2008 to 31 March 2010). The first low-flow benefits following installation of the improved facilities were expected during the summer of 2008.

### *2.4.2 Results*

Overall results from the Blackwater subcatchment (Fig. 2.9) do not provide a firm basis for rejecting the $H_0$. On a more local scale, results for the subcatchment in Co. Armagh, where there was little change in the density of systems, indicate some positive effects on water quality following improvements to septic tank systems. However, in other parts of the study area no significant improvements in water quality were evident. The latter may reflect overall increases in the total number of septic tank systems due to new builds of rural housing during the survey period (land use change may also have influenced the results). This finding is of particular concern, given that the study period also included the implementation of other efforts to mitigate P inputs, notably farm yard improvements and extensive additional fencing to restrict the access of livestock to surface water bodies.

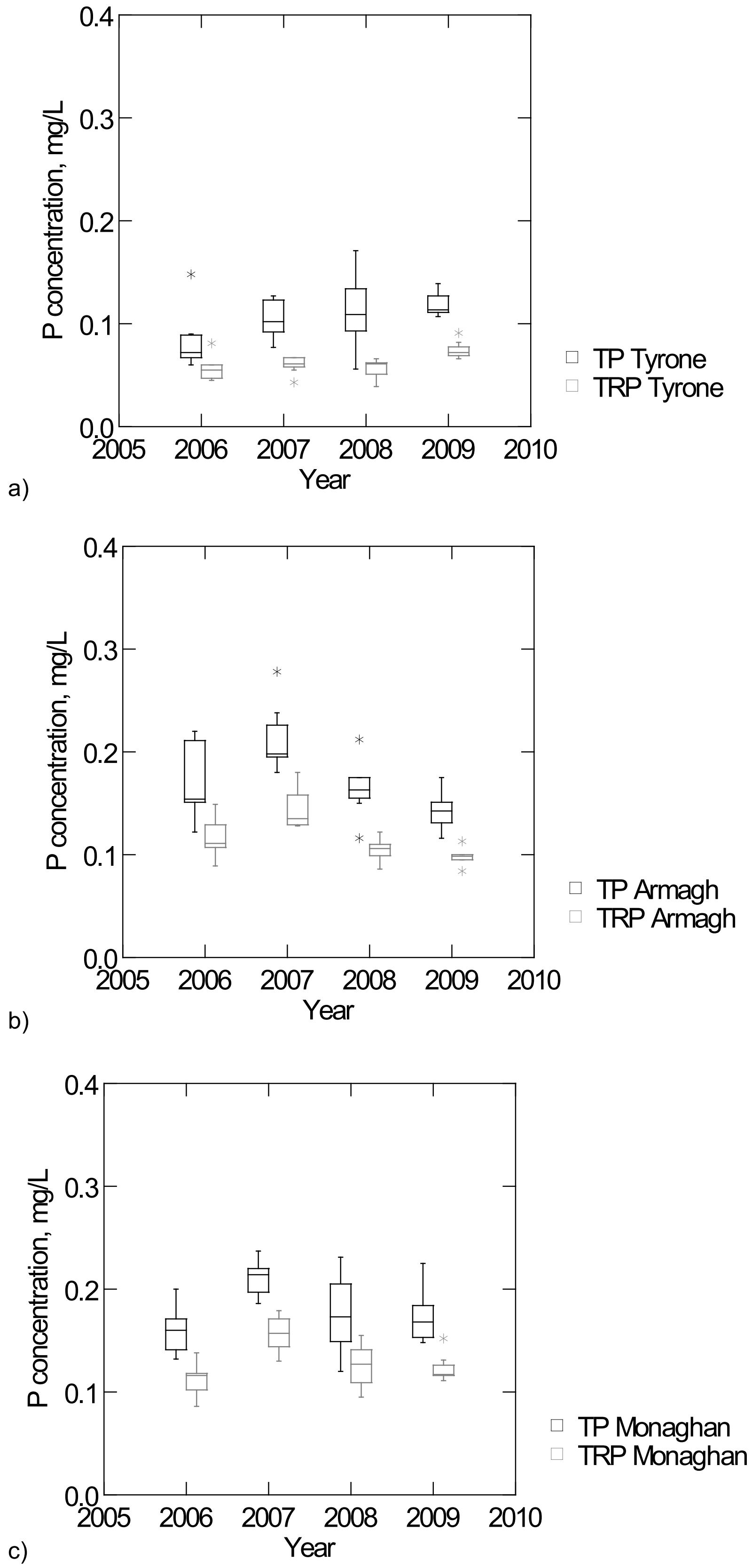


**Figure 2.9. Box-whisker plots showing the distribution of average concentrations of phosphorus (P), total phosphorus (TP) and total reactive phosphorus (TRP) in each 0.01m3 s$^{-1}$ bin of discharge below 0.09m3 s$^{-1}$ in the three study subcatchments in the Blackwater catchment (Work Package 4): Co. Tyrone (a) indicates a slight but significant increase in concentration over the period; Co. Armagh a significant decrease (b); and Co. Monaghan a small (but insignificant) increase (c).**

# 3 Synthesis of EFFECT Findings

Persistent and considerable uncertainty regarding the suitability and effectiveness of some POMs (e.g. Mayes and Codling, 2009; Schulte *et al.*, 2010) provided the context for the EFFECT project. Specifically, and focusing on the Irish Ecoregion, EFFECT sought to determine the extent to which the implementation of measures aimed at reducing nutrient inputs affected the quality of rivers, streams and lakes. This chapter synthesises the findings of the project.

The impact of different riparian vegetation types on the ecological quality of streams draining upland peatland catchments in NI was assessed in WP3. Coniferous forest plantations were established in much of the study catchments between the 1960s and 1980s, although felling operations have recently reduced their extent. Replanting has taken place, and riparian vegetation is being restructured in an attempt to improve and protect stream water quality. However, the extent to which water quality in headwater streams is a function of riparian vegetation or of plant cover over larger areas – and hence the efficacy of bankside management – is debatable: riparian vegetation is only one factor affecting nutrient loads (Pinay *et al.,* 1992) and its influence is likely to be mediated by the proximity of sources and density of channels (Norris, 1993). Results generated in WP3 suggest that riparian vegetation did not have a major influence on macrophyte and invertebrate composition at the study sites. More influential catchment-scale factors appeared to be exerted through their effects on water chemistry.

The apparent ineffectiveness of POMs aimed at improving water quality in headwater streams in afforested peatland catchments may be a consequence of their relatively limited extent (a relatively narrow riparian zone) when compared with the landscape-wide scale of the pressures. The dependence of POM effectiveness on scale and density were also factors impacting P reduction attempts assessed in WP4, in which the strategic replacement of *c.* 10% of septic tank systems in part of the Blackwater catchment in Co. Armagh led to reduced P concentrations during ecologically important low flows. Despite this improvement, overall P loads were largely similar before and after replacement facilities came online, largely because of the prevalence of P from diffuse sources. In other parts of the Blackwater catchment covered by the study, the replacement and upgrading of septic tank systems had no significant P concentration effects, despite the implementation of additional POMs aimed at reducing P inputs from point and diffuse sources. Throughout the Blackwater catchment an increased density of septic tank systems, partly associated with the construction of new housing, was an important confounding influence on attempts to mitigate concentrations of P released to water bodies. Land use change in the subcatchments during the study period may also have influenced results. The combined effects of these confounding factors may have largely outweighed any benefits of POM implementation.

Part of the difficulty in assessing the effectiveness of implementation of POMs stems from the need to ensure comparability of data (Zeitsch *et al.*, 2009). For example, linking P loads to livestock numbers directly over several years of data runs the risk of failing to account for dramatic changes in the composition of feed over time, and therefore the quantity and quality of material excreted. Thus, while livestock numbers may show little if any change over the last *c*. 20 years, their productivity has risen sharply, and with it the release of increased levels of nutrients per animal – potentially overcoming any benefits of reductions in fertiliser applications (Maguire *et al.*, 2009).

Other reasons more closely relate to the varying pathways and processes linking P released from point and diffuse sources with aquatic effects of enrichment. Unravelling this complexity is fundamental to understanding apparent lags between POM implementation and improvements in CWQ and BWQ. Transfer pathways of P from diffuse and point sources to water bodies are different and have a contrasting relationship with flow conditions (Chapman, 1996). Point sources of P deliver relatively constant loads during the hydrological cycle of a river and are independent of flow. Conversely, diffuse sources are highly dependent on flow for transfer from land to water (Edwards and Withers, 2007); an increase in P concentration with flow is caused by a combination of nutrients washed off from the land in runoff, river bed remobilisation and erosion processes (Best *et al.*, 1997). During large precipitation events, the diffuse component of P loads can dominate instantaneous river loads (Webb *et al.*, 1997; Tunney *et al.*, 2000; Jordan *et al.*, 2005a and b). In the intervals between high-flow events, P concentrations are low when runoff pathways are disconnected and/or when point sources are absent or when a sufficient dilution factor exists. P concentrations may be high during low flows when a point source is operating (Arnscheidt *et al.*, 2007; Douglas *et al.*, 2007). As high P concentrations are likely to maintain an enhanced trophic state, aquatic ecological health may be threatened, particularly if high concentrations of P are in the form of MRP and coincide with the growing season (Bowes *et al.*, 2010). Furthermore, the extent of overlap between periods of oversupply of P and biological demand is critical in determining the relative importance of different sources and their aquatic ecological impacts (Edwards and Withers, 2007).

Exports of P from a catchment are primarily a function of land use (Wood *et al.*, 2005), soil conditions (Jordan *et al.*, 2005b), the connectivity between hydrology and catchment management (Foy *et al.*, 1995; Foy and Lennox, 2000; Ulén and Jacobsson, 2005) and P loading (Donohue *et al.*, 2006). A variety of approaches has been applied to determine the sources and transfer dynamics of P loads (May *et al.*, 2001; Bowes *et al.*, 2005; Withers and Jarvie, 2008). One – the source approach – estimates P loads before entry into a watercourse, using data from the monitoring of point contributors (Foy *et al.*, 1995). In the P export coefficient approach, coefficients are applied in order to estimate P loading using catchment attributes, such as population density or the extent of different land covers and land uses (Johnes, 1996). Both methods require detailed information on catchment activities and physical attributes, but they are unable to accommodate random pollution events or the effects of annual hydroclimatic variations. Using an approach based on water quality monitoring to calculate P load (Quilbé *et al.*, 2006) can also be problematic, because such an approach does not usually differentiate between sources.

Work Package 1 findings provided insight on the environmental influences on P transfers to and impacts upon surface water bodies, in-lake P dynamics, and the basis for deterministic geospatial models for use in predicting areas of high P vulnerability. Thus, the main determinants of P concentrations in rivers in the study catchments from across the Irish Ecoregion were found to comprise, in order of importance, mean catchment slope, degree of soil saturation, human population density, soil P runoff risk and extent of pasture. Exports of P delivered to rivers, and the relationships between catchment attributes and P, were also found to vary seasonally., The difficulty in distinguishing the possible sources of P in water-quality monitoring data was addressed through the development and application of a LAM that separated contributions from point and diffuse loads through the examination of the relationship between P concentration and discharge (Greene *et al.*, 2011). The LAM deployed in WP1 also facilitated the identification of subcatchments draining to Lough Sheelin that contributed disproportionately high loads of P annually, and therefore where effective implementation of P mitigation measures may be most urgently needed.

Constraints to biological recovery are also a factor governing the recovery of water bodies (Bond and Lake, 2003; Langford *et al.*, 2009). Climate change may exacerbate pollution and alter species and ecosystem level responses, and may also confound the efficacy of certain measures (Jennings *et al.*, 2009). For example closed seasons for slurry spreading are designed to coincide with the times of the year when high soil moisture conditions are likely to prevail, thereby heightening risks of P losses. Altered and more variable weather patterns as a consequence of climate change may therefore undermine such measures. Moreover, impacts of P in rivers and lakes will vary according to the time of year, the nature of the water body and the organism(s) of interest (Brabec *et al.,* 2004; Hilton *et al.,* 2006; O'Driscoll *et al.,* 2006; Schippers *et al.,* 2006; Buenau *et al.,* 2007; Ibelings *et al.,* 2007; Jeppesen *et al.,* 2007; Ulén *et al.,* 2007). Biological recovery may be limited simply by a shortage of colonising taxa (Sundermann *et al.,* 2011b), although this was not thought to be a factor in either the Upper Bann or Colbrooke catchments (WP2) where larger numbers of taxa associated with lower levels of organic pollution were often found in the main river channels when compared with headwater streams. Research carried out as part of WP2, however, revealed a potentially important role for hydromorphology, and in particular the impediment of biological recovery in streams showing improvements in CWQ because channelisation, siltation and the presence of biofilms reduced habitat opportunities. A role for altered hydromorphology in constraining ecological recovery has also recently been highlighted in an extensive study of restoration impacts in rivers in Germany (Sundermann et al., 2011a). Another constraint on recovery, following implementation of POMs aimed at mitigating P effects, is an increased density of rural housing and associated waste-disposal infrastructure. As suggested in the results for WP4, the effects of such changes in the density of housing in rural areas can easily offset any improvements in water quality that might be expected from the implementation of POMs targeting previously established pollutant sources.

Incomplete knowledge of how invasive species interact with native biota may also obscure relationships between improved CWQ and aquatic biotic response. Invasive species may potentially restrict recovery (Kelly *et al.*, 2003) and/or exploit restoration measures better than native species (Zedler, 2000). Zebra mussels were established as an invasive species in Lough Sheelin by 2004. Results presented here as part of WP1 indicate that zebra mussel activity has likely been responsible for increased TP concentrations in the lake as a result of heightened internal loading due to an increase in the mobilisation of sediment-bound P, thereby confounding the effects of apparently successful attempts to reduce external TP inputs. The effects of zebra mussel establishment on the availability of P to other organisms are not straightforward, however, and may depend on the trophic status of a lake when invasion commenced (Qualls *et al.*, 2007; Vanderploeg *et al.*, 2010), with availability tending to increase where P levels are already high (Mellina *et al.*, 1995; Vanderploeg *et al.*, 2001). Other factors confounding the relationship between zebra mussel establishment and water quality include the number and size of individuals comprising the established population (Idrisi *et al.*, 2001), the degree of stratification (Higgins *et al.*, 2011) and the presence of predators, parasites and ecological competitors (Molloy *et al.*, 2001). Rather than impacting biological recovery through environmental modification, the invasive amphipod *Gammarus pulex* may through predation (Kelly *et al.,* 2003; Dick, 2008) have a more direct effect by restricting the recolonisation of streams in the Upper Bann (WP2) by pollution-sensitive invertebrate taxa, including the native *Gammarus duebeni celticus*.

One-off and episodic pollution events may also restrict recovery by periodically extinguishing biota (Kowalik *et al.*, 2007). Since the chemical longevity of such occurrences can be short lived and missed by standard surveillance monitoring, attributing biological impacts is particularly difficult. This is especially true for agricultural effluents associated with silage and slurry spreading, which, although discharged intermittently, have a high pollution capacity and potentially cumulative effects (Seager and Maltby, 1989; Foy *et al.*, 2001). Enhanced pollution events of short duration, linked to anomalous levels of rainfall and to the accidental or deliberate release of pollutants, may have limited the efficacy of POMs in the Lough Sheelin (WP1) and Upper Bann and Blackwater (WPs 2 and 4) catchments. Moreover, anomalously high levels of rainfall throughout much of Ireland in recent years – and in particular a higher than normal frequency of storm events – may, by increasing inputs of pollutants from diffuse sources, have adversely impacted water quality in agricultural catchments in many parts of the Irish Ecoregion (McGarrigle *et al.,* 2010).

While the occurrence and aquatic biotic effects of one-off and episodic pollution events present a strong argument for high-frequency surveillance monitoring (Jordan et al., 2007), the cost of carrying out the latter over a large area is likely to prove prohibitive. The use of models based on existing datasets, such as the LAM and geospatial model developed in the current study, can facilitate a cost-effective deployment of high-frequency surveillance monitoring by identifying subcatchments and water bodies at high risk.

Other potential reasons why the implementation of POMs have had limited success, and which could not be investigated in the EFFECT project because of resource constraints, include the delayed, incomplete or uneven application of measures. Delays may, in some cases, stem from the challenges their implementation pose for intensive farming practices (Oenema, 2004). The possibility that the implementation of some measures on paper may not have been carried through into practice in a consistent way is increased where the focus of regulation is widely dispersed and relatively remote, as is the case with nutrient sources in many rural catchments in RoI and NI. Moreover, the EFFECT project was largely focused on poorly drained soils, which in the Irish Ecoregion tend to be associated with the most acute water quality problems in agricultural catchments. As a consequence, the findings, especially regarding the factors that appear to influence the efficacy of POMs, may not be representative for the entire Irish Ecoregion.

# 4 Conclusions, Implications and Recommendations

There remains a relative paucity of studies of the impacts of implementation of POMs aimed at mitigating effects of nutrients, notably P, on surface bodies of freshwater and over ecologically meaningful timescales, both generally and in the Irish Ecoregion specifically. Results presented in this report suggest that many existing POMs have proven or are proving ineffective in raising BWQ and restoring ecological functioning.

The somewhat disappointing results following the implementation of POMs in the Irish Ecoregion indicate a need for an improved understanding of the processes influencing the production and transport of pollutants – particularly from diffuse agricultural and forestry sources, and their impacts on aquatic ecosystems. Based on findings presented in this report and supported in the published literature, the effectiveness of POMs is determined by a range of environmental and human factors and appears to be scale, density and time dependent. These factors and dependencies deserve to be taken into consideration when designing and implementing cost-effective measures aimed at mitigating the biological and ecological effects of nutrient enrichment. There are positives, however. Some results demonstrate that improvements are possible, even where measures are implemented voluntarily. Furthermore, cases exist of early signs of biological improvement in association with changes in CWQ following the implementation of POMs. Moreover, known relationships between environmental and socio-economic factors and the effectiveness of POMs can be used as a basis for identifying subcatchments and bodies of water that are likely to be particularly vulnerable to P impairment and slow to recover following the implementation of measures aimed at reducing P loads.

The following implications arise from the results of the EFFECT project:

- Findings that both point and diffuse sources of P hindered the effectiveness of POMs at different stages during the hydrological cycle provide a powerful argument for long-term monitoring that fully samples the flow range of rivers, especially in subcatchments with multiple potential sources of nutrients and sensitive environmental conditions (e.g. impermeable soils/high densities of septic tanks). The LAM constructed in this research can potentially help here.
- The contribution of septic tanks to high P concentrations in rivers, particularly as constant point sources during low flows in the summer months, was a recurring theme in the EFFECT project and has relevance to many of the basic measures implemented under the WFD to enact national legislation for domestic septic discharges. Furthermore, P discharges from septic tanks may also confound the effectiveness of agriculturally focused POMs, such as those prescribed under the Nitrates Directive.
- The EPA has proposed to estimate lake-by-lake P loading standards through Vollenweider-type modelling of the response of lakes to P inputs. However, data from this study highlight that there are difficulties in using these models for lakes where the P cycle is not fully understood or has recently been altered – for example, by the activities of invasive species. To evaluate the success of POMs in all lakes, separate standards and new models need to be developed for invaded and non-invaded systems.
- Farm-based measures aimed at reducing nutrient loads to rivers have shown divergent outcomes. Farm pollution has been reduced and metrics of CWQ have improved. Catchment land use changes have resulted in marked reductions in $NO_3$ concentrations and, to a lesser extent, P concentrations, particularly on those farms where nutrient use was targeted specifically. Improvements in BWQ based on macroinvertebrate composition have not followed reduced nutrient concentrations, however.
- While low-order streams draining small catchments do not require monitoring under the WFD requirements, first-order headwater streams can contribute large proportions of water volume

and N to higher-order streams. Moreover, a high proportion of stream length in a catchment is made up of streams < 10 m width. Low-order streams – and activities such as commercial forestry that are frequently associated with headwater catchments in the Irish Ecoregion – can thus have a major influence on water quality further downstream.

- The composition of invertebrate and macrophyte assemblages and invertebrate community structure and functioning in streams in commercially forested catchments are more strongly correlated with landscape scale factors than with riparian zone characteristics.
- Improvements in low-flow water quality due to the replacement of a sample of defective septic tank systems yielded mixed results, most likely due to inter-subcatchment differences in the siting and functioning of septic tank systems and drainage and concurrent increases in septic system density from new-builds. Land use change in the study subcatchments may also have impacted effectiveness of improvements to septic tank systems and other POMs introduced during the time period of the study.
- Monitoring by the EPA in the RoI currently targets a sample of sites that are regarded as representative, leaving many rivers and lakes unmonitored. However, under the WFD, regulatory organisations such as the EPA are obliged in 2015 to submit an up-to-date report of the chemical and ecological status of both monitored and unmonitored water bodies. Geospatial models, such as the RK model developed in the current study, can be used to identify water bodies that are vulnerable to external loadings of P, and this information can be used as a guide to their current chemical and ecological status.
- Many POMs have only relatively recently been implemented, and in some cases implementation is likely to have been partial at best. Given that water bodies in the Irish Ecoregion have been profoundly impacted by poor CWQ over several decades, expecting improvements in BWQ and ecological functioning in rivers and lakes so soon after implementation of POMs is perhaps overly optimistic.

The WFD is mainly concerned with relatively large bodies of water that are subject to a variety of different anthropogenic impacts, thus complicating the detection of strict responses to agricultural measures (Statzner and Beche, 2010). The third (and final) National Implementation Report states specifically that local authorities should identify and rectify the causes of decline in high-quality waters (Clenaghan *et al.,* 2005). Models developed in WP1 could help achieve the first of these. Furthermore, and as highlighted in this report, rectification of decline will involve an understanding of conditions at the subcatchment level, and take into account land use in the most hydrologically active parts. Some improvements to measures are recommended as a result of investigations carried out through the EFFECT project. Relatively narrow strips of riparian vegetation are clearly ineffective buffers against much larger-scale changes in catchment conditions. Moreover, some rivers and lakes have been profoundly modified hydromorphologically and through the activities of invasive taxa, and in these cases biological recovery is bound to lag behind changes in chemical water quality. Hysteresis can also be expected in some cases (Lepori *et al.,* 2005). As a consequence of these factors, the pathway of biological recovery of a heavily modified river or lake could potentially be very different from the expected.

Future research should aim to place greater emphasis on relatively well-drained soils than was possible in the current project, and focus on understanding the links between sources, pathways, sinks and biological impacts of P and other nutrients, and the factors that potentially can modify pollution effects. Results of monitoring now being conducted for the WFD may be able to make a useful contribution here in refining and improving the models developed through the EFFECT project. Moreover, attention should be paid to the degree that POMs have been and are being implemented.

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# Acronyms and Annotations

| | |
|---|---|
| CWQ | Chemical water quality |
| ASPT | Average score per taxon |
| BMWP | Biological Monitoring Working Party |
| BWQ | Biological water quality |
| C | Carbon |
| EFFECT | Effective Framework For assessing aquatic ECosysTem responses to implementation of the Phosphorous Regulations |
| fwMCs | Flow weighted mean concentrations |
| fwMRP | Flow-weighted P data |
| FFG | Functional feeding group |
| $H_o$ | null hypothesis |
| NI | Northern Ireland |
| LFA | Less favoured area |
| LAM | Load apportionment model |
| MRP | Molybdate reactive phosphorus |
| NAP | Nitrates Action Programme |
| EU | European Union |
| P | Phosphorus |
| P Regs | Phosphorus Regulations |
| PCA | Principal Components Analysis |
| POMs | Programmes of Measures |
| PAR | Reduction in light intensity |
| RBD | River basin district |
| RMC | Riparian management categories |
| ROI | Republic of Ireland |
| SI | Stable isotope |
| DARD | Department of Agriculture and Rural Development (Northern Ireland) |
| TP | Total phosphorus |
| WWTPs | Waste water treatment plants |
| WFD | Water Framework Directive |

# An Ghníomhaireacht um Chaomhnú Comhshaoil

Is í an Gníomhaireacht um Chaomhnú Comhshaoil (EPA) comhlachta reachtúil a chosnaíonn an comhshaol do mhuintir na tíre go léir. Rialaímid agus déanaimid maoirsiú ar ghníomhaíochtaí a d'fhéadfadh truailliú a chruthú murach sin. Cinntímid go bhfuil eolas cruinn ann ar threochtaí comhshaoil ionas go nglactar aon chéim is gá. Is iad na príomh-nithe a bhfuilimid gníomhach leo ná comhshaol na hÉireann a chosaint agus cinntiú go bhfuil forbairt inbhuanaithe.

Is comhlacht poiblí neamhspleách í an Ghníomhaireacht um Chaomhnú Comhshaoil (EPA) a bunaíodh i mí Iúil 1993 faoin Acht fán nGníomhaireacht um Chaomhnú Comhshaoil 1992. Ó thaobh an Rialtais, is í an Roinn Comhshaoil, Pobal agus Rialtais Áitiúil.

## ÁR bhFREAGRACHTAÍ

### CEADÚNÚ

Bíonn ceadúnais á n-eisiúint againn i gcomhair na nithe seo a leanas chun a chinntiú nach mbíonn astuithe uathu ag cur sláinte an phobail ná an comhshaol i mbaol:

- áiseanna dramhaíola (m.sh., líonadh talún, loisceoirí, stáisiúin aistrithe dramhaíola);
- gníomhaíochtaí tionsclaíocha ar scála mór (m.sh., déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta);
- diantalmhaíocht;
- úsáid faoi shrian agus scaoileadh smachtaithe Orgánach Géinathraithe (GMO);
- mór-áiseanna stórais peitreail;
- scardadh dramhuisce.

### FEIDHMIÚ COMHSHAOIL NÁISIÚNTA

- Stiúradh os cionn 2,000 iniúchadh agus cigireacht de áiseanna a fuair ceadúnas ón nGníomhaireacht gach bliain.
- Maoirsiú freagrachtaí cosanta comhshaoil údarás áitiúla thar sé earnáil - aer, fuaim, dramhaíl, dramhuisce agus caighdeán uisce.
- Obair le húdaráis áitiúla agus leis na Gardaí chun stop a chur le gníomhaíocht mhídhleathach dramhaíola trí comhordú a dhéanamh ar líonra forfheidhmithe náisiúnta, díriú isteach ar chiontóirí, stiúradh fiosrúcháin agus maoirsiú leigheas na bhfadhbanna.
- An dlí a chur orthu siúd a bhriseann dlí comhshaoil agus a dhéanann dochar don chomhshaol mar thoradh ar a ngníomhaíochtaí.

### MONATÓIREACHT, ANAILÍS AGUS TUAIRISCIÚ AR AN GCOMHSHAOL

- Monatóireacht ar chaighdeán aeir agus caighdeáin aibhneacha, locha, uiscí taoide agus uiscí talaimh; leibhéil agus sruth aibhneacha a thomhas.
- Tuairisciú neamhspleách chun cabhrú le rialtais náisiúnta agus áitiúla cinntí a dhéanamh.

### RIALÚ ASTUITHE GÁIS CEAPTHA TEASA NA HÉIREANN

- Cainníochtú astuithe gáis ceaptha teasa na hÉireann i gcomhthéacs ár dtiomantas Kyoto.
- Cur i bhfeidhm na Treorach um Thrádáil Astuithe, a bhfuil baint aige le hos cionn 100 cuideachta atá ina mór-ghineadóirí dé-ocsaíd charbóin in Éirinn.

### TAIGHDE AGUS FORBAIRT COMHSHAOIL

- Taighde ar shaincheisteanna comhshaoil a chomhordú (cosúil le caighdéan aeir agus uisce, athrú aeráide, bithéagsúlacht, teicneolaíochtaí comhshaoil).

### MEASÚNÚ STRAITÉISEACH COMHSHAOIL

- Ag déanamh measúnú ar thionchar phleananna agus chláracha ar chomhshaol na hÉireann (cosúil le pleananna bainistíochta dramhaíola agus forbartha).

### PLEANÁIL, OIDEACHAS AGUS TREOIR CHOMHSHAOIL

- Treoir a thabhairt don phobal agus do thionscal ar cheisteanna comhshaoil éagsúla (m.sh., iarratais ar cheadúnais, seachaint dramhaíola agus rialacháin chomhshaoil).
- Eolas níos fearr ar an gcomhshaol a scaipeadh (trí cláracha teilifíse comhshaoil agus pacáistí acmhainne do bhunscoileanna agus do mheánscoileanna).

### BAINISTÍOCHT DRAMHAÍOLA FHORGHNÍOMHACH

- Cur chun cinn seachaint agus laghdú dramhaíola trí chomhordú An Chláir Náisiúnta um Chosc Dramhaíola, lena n-áirítear cur i bhfeidhm na dTionscnamh Freagrachta Táirgeoirí.
- Cur i bhfeidhm Rialachán ar nós na treoracha maidir le Trealamh Leictreach agus Leictreonach Caite agus le Srianadh Substaintí Guaiseacha agus substaintí a dhéanann ídiú ar an gcrios ózóin.
- Plean Náisiúnta Bainistíochta um Dramhaíl Ghuaiseach a fhorbairt chun dramhaíl ghuaiseach a sheachaint agus a bhainistiú.

### STRUCHTÚR NA GNÍOMHAIREACHTA

Bunaíodh an Ghníomhaireacht i 1993 chun comhshaol na hÉireann a chosaint. Tá an eagraíocht á bhainistiú ag Bord lánaimseartha, ar a bhfuil Príomhstiúrthóir agus ceithre Stiúrthóir.

Tá obair na Gníomhaireachta ar siúl trí ceithre Oifig:

- An Oifig Aeráide, Ceadúnaithe agus Úsáide Acmhainní
- An Oifig um Fhorfheidhmiúchán Comhshaoil
- An Oifig um Measúnacht Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáide

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag ball air agus tagann siad le chéile cúpla uair in aghaidh na bliana le plé a dhéanamh ar cheisteanna ar ábhar imní iad agus le comhairle a thabhairt don Bhord.

**Science, Technology, Research and Innovation for the Environment (STRIVE) 2007-2013**

The Science, Technology, Research and Innovation for the Environment (STRIVE) programme covers the period 2007 to 2013.

The programme comprises three key measures: Sustainable Development, Cleaner Production and Environmental Technologies, and A Healthy Environment; together with two supporting measures: EPA Environmental Research Centre (ERC) and Capacity & Capability Building. The seven principal thematic areas for the programme are Climate Change; Waste, Resource Management and Chemicals; Water Quality and the Aquatic Environment; Air Quality, Atmospheric Deposition and Noise; Impacts on Biodiversity; Soils and Land-use; and Socio-economic Considerations. In addition, other emerging issues will be addressed as the need arises.

The funding for the programme (approximately €100 million) comes from the Environmental Research Sub-Programme of the National Development Plan (NDP), the Inter-Departmental Committee for the Strategy for Science, Technology and Innovation (IDC-SSTI); and EPA core funding and co-funding by economic sectors.

The EPA has a statutory role to co-ordinate environmental research in Ireland and is organising and administering the STRIVE programme on behalf of the Department of the Environment, Heritage and Local Government.

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