The Environmental Protection Agency

RIVER SEDIMENT STUDIES IN RELATION TO JUVENILE PEARL MUSSELS AND SALMONIDS

Environmental Protection Agency
An Ghníomhaireacht um Chaomhnú Comhshaoil

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.

OUR RESPONSIBILITIES

LICENSING

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

- waste facilities (e.g., landfills, incinerators, waste transfer stations);
- large scale industrial activities (e.g., pharmaceutical manufacturing, cement manufacturing, power plants);
- intensive agriculture;
- the contained use and controlled release of Genetically Modified Organisms (GMOs);
- large petrol storage facilities.
- waste water discharges

NATIONAL ENVIRONMENTAL ENFORCEMENT

- Conducting over 2,000 audits and inspections of EPA licensed facilities every year.
- Overseeing local authorities' environmental protection responsibilities in the areas of - air, noise, waste, waste-water and water quality.
- Working with local authorities and the Gardaí to stamp out illegal waste activity by co-ordinating a national enforcement network, targeting offenders, conducting investigations and overseeing remediation.
- Prosecuting those who flout environmental law and damage the environment as a result of their actions.

MONITORING, ANALYSING AND REPORTING ON THE ENVIRONMENT

- Monitoring air quality and the quality of rivers, lakes, tidal waters and ground waters; measuring water levels and river flows.
- Independent reporting to inform decision making by national and local government.

REGULATING IRELAND'S GREENHOUSE GAS EMISSIONS

- Quantifying Ireland's emissions of greenhouse gases in the context of our Kyoto commitments.
- Implementing the Emissions Trading Directive, involving over 100 companies who are major generators of carbon dioxide in Ireland.

ENVIRONMENTAL RESEARCH AND DEVELOPMENT

- Co-ordinating research on environmental issues (including air and water quality, climate change, biodiversity, environmental technologies).

STRATEGIC ENVIRONMENTAL ASSESSMENT

- Assessing the impact of plans and programmes on the Irish environment (such as waste management and development plans).

ENVIRONMENTAL PLANNING, EDUCATION AND GUIDANCE

- Providing guidance to the public and to industry on various environmental topics (including licence applications, waste prevention and environmental regulations).
- Generating greater environmental awareness (through environmental television programmes and primary and secondary schools' resource packs).

PROACTIVE WASTE MANAGEMENT

- Promoting waste prevention and minimisation projects through the co-ordination of the National Waste Prevention Programme, including input into the implementation of Producer Responsibility Initiatives.
- Enforcing Regulations such as Waste Electrical and Electronic Equipment (WEEE) and Restriction of Hazardous Substances (RoHS) and substances that deplete the ozone layer.
- Developing a National Hazardous Waste Management Plan to prevent and manage hazardous waste.

MANAGEMENT AND STRUCTURE OF THE EPA

The organisation is managed by a full time Board, consisting of a Director General and four Directors.

The work of the EPA is carried out across four offices:

- 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.

RIVER SEDIMENT STUDIES IN RELATION TO JUVENILE PEARL MUSSELS AND SALMONIDS

NIAMH WALSH, MICHAEL NEILL
AND JOHN LUCEY

Aquatic Environment
Office Of Environmental Assessment

Environmental Protection Agency
An Ghníomhaireacht Um Chaomhnú Comhshaoil
Seville Lodge
Callan Road
Kilkenny

July 2012
ISBN 978-1-84095-457-9

Printed On Recycled Paper

FOREWORD

Sediment transport has increased over the last century in the rivers of northern Europe. In Ireland, many of our rivers now carry increased amounts of fine sediment, as a consequence of more intensive land use, which can lead to siltation of their beds. This fine sediment can adversely affect juvenile pearl mussels and salmonid fishes, buried in the sediment, which depend on a plentiful supply of oxygen to their habitat.

The studies reported on here, relating to siltation, had as their main aims to establish if river substratum dissolved oxygen could be measured directly in the field and to investigate siltation processes, particularly in relation to juvenile salmonid and pearl-mussel habitats.

Both the salmon and pearl mussel are protected under the **Habitats Directive**. The species' protection requirements are the central tool for bringing protected species and habitats into a favourable conservation status. The aim of the **Water Framework Directive (WFD)** is to establish a framework for the protection of waters, with the objective to reach good status by 2015. Both the Natura directive and the WFD aim at ensuring healthy aquatic ecosystems, while at the same time ensuring a balance between water/nature protection and the sustainable use of natural resources. Implementation of measures under the WFD will support the objectives of the Habitats Directive.

While the precise requirements of the freshwater pearl mussel (*Margaritifera margaritifera*) are not fully established, it is known that siltation, leading to depleted oxygen, adversely affects the juveniles and their host fish which in Ireland are salmon and trout (salmonids). Until now, the direct measurement of oxygen has not been carried out in these species habitats. Rather, proxy or indirect measurements have been employed.

The late Michael Neill, who led our laboratory in Kilkenny and initiated the studies, believed that dissolved oxygen could be successfully measured in river sediments (interstitial water) and the main outcome of the project vindicates this view. I wish to record our appreciation for the work of this study team, which began with Michael and was continued by Niamh Walsh and our Aquatic Environment colleague in Kilkenny, John Lucey.

The report also concludes that while some siltation of rivers in south-east Ireland is occurring, the nitrate levels in some are more of an immediate problem for sustainable pearl mussel populations. The management of nitrates is an ongoing challenge in the region.

Another part of the studies, relating to turbidity and suspended solids, did not have such clear-cut conclusions. It yielded no clear relationship between turbidity and suspended solids in the River Nore, with the suspicion that colour had a strong influence on turbidity results.

The Agency has commissioned, under the STRIVE programme, a large-scale project *Measurement of silt flux in rivers and benefits of enhancement measures* (SILTFLUX). Hopefully, that project will help to overcome some of the problems in using turbidity to predict suspended solids concentrations, so that standards for suspended solids flux and concentrations, for the protection of sensitive catchments in Ireland, can be established.

Micheál Ó Cinnéide
Director
Office of Environmental Assessment

PROJECT STATEMENT

This project was initiated by Michael Neill (1948-2010) under the FÁS Graduate Work Placement Programme Scheme. Following his untimely death the project was supervised by John Lucey. The report is dedicated to Michael's memory.

ACKNOWLEDGEMENTS

The authors wish to gratefully acknowledge the following EPA colleagues in Kilkenny for their help in field surveys and laboratory assistance: Diarmuid Berry, Albert Curran, Eugene Halley, Eamon Holohan, Christopher Murray, Joseph Reilly and Adrienne Turner. Also thanks to Bernadette Cuddihy for ordering equipment for the project and to Catherine O'Keefe for literature retrieval as well as to Rebecca Quinn (EPA Dublin) and OPW for additional flow data. Special thanks are due to Áine O'Connor, of the National Parks & Wildlife Service (NPWS), for kindly lending additional Sonde water quality data loggers and to Evelyn Moorkens (Evelyn Moorkens & Associates) for advice on using redox potential instrumentation. The provision of rainfall data by Noreen Brennan (Met Éireann) is greatly appreciated. Finally, Inland Fisheries Ireland (IFI) teams of Tony Byrne/Noel Power and Declan Cullagh/Michael Byron, are thanked for providing raw data from electric-fishing surveys.

TABLE OF CONTENTS

Foreword	iv
Project Statement	v
Acknowledgements	v
Summary	vii
Introduction	1
Materials and methods	6
Study areas	10
Results	15
Discussion	50
References	57

SUMMARY

Key words: interstitial water, siltation/sedimentation, suspended solids, turbidity, colour, conductivity, dissolved oxygen, redox potential, nitrate, nitrite, phosphate, pearl mussels, salmonids

The main aims of the study were essentially to establish if river substratum dissolved oxygen could be measured directly in the field and to investigate siltation processes particularly in relation to juvenile salmonid and pearl-mussel habitats. The field studies were carried out on pearl-mussel rivers in the south-east of Ireland with emphasis on the River Nore (Cos Laois/Kilkenny) which is home to the last surviving population of the hard-water form (ecophenotype) of *Margaritifera margaritifera*.

The study had effectively two linked phases with the following objectives:

1. To measure silt accumulation along the River Nore during base flow conditions and examine the relationship between turbidity/suspended solids measurements;
2. To test a method of directly measuring dissolved oxygen in river substrata which to date has been done by proxy, i.e. by taking redox potential readings to indicate oxidation of sediments, or by extracting water and then analysing the samples.

For the siltation study, sediment traps were deployed for a four-week period at each of the two sampling sites on the River Nore. The mean estimates for sedimentation rates for fines (silt and clay) at the two sites, with average flows of 0.67 and 3.20 m^3/S, gave very similar results: 8.73 $g/m^2/d$ at Kilbricken and 8.83 $g/m^2/d$ at Ballyragget. These rates were higher than those estimated for another pearl-mussel river, the Dawros, in Co Galway and would appear much higher than accumulation of sediment in some Swedish *Margaritifera* rivers. The two Irish studies employed the same materials and methods. There is a need for standardization of methodology so that various studies can be judiciously compared.

Turbidity and suspended solids measurements were carried out to investigate if there was a relationship between the two parameters to establish if the more easily measured turbidity could be used as a surrogate for suspended solids. A close correlation was found during a preliminary trial (r^2=0.9696) at Kilbricken during variable flow conditions. However, no association was shown between turbidity and suspended solids during the subsequent four-week study at that site and at Ballyragget as well as at an intermediate location (New Bridge) in lower steady-flow conditions. Thus, the study yielded no clear relationship between turbidity and suspended solids with an r^2 value of 0.0146 for all samples. At the uppermost site the indications were that water colour was strongly influencing turbidity values under certain conditions. It is concluded that if such an association does exist in the River Nore, then studies under a wide variety of hydrographical conditions would be necessary. A statistically significant inverse relationship between colour and conductivity was apparent in the River Nore which became stronger with passage downstream.

Turbidity has been used as a proxy for sedimentation in other countries but given the high natural colour of some Irish river sites it is unlikely to yield meaningful results in these. Mean turbidity values (n=17) over the August September period of 2010, at a time of little precipitation and run-off, were 7.7, 1.4 and 0.5 NTU at the three Nore sites going downstream. A mean threshold turbidity level of 1.0-1.9 NTU has been extrapolated for Swedish pearl mussel-recruitment rivers.

A level of 30 mg/l of suspended solids has been given as the limit of tolerance by adult pearl mussels. The Freshwater Fish Directive includes an annual average guideline standard of 25 mg/l suspended solids but will be repealed in 2013 by the Water Framework Directive which does not

specify an environmental objective for this parameter. Suspended sediment (suspended solids) results for designated salmonid rivers in the south-east, which also support *M. margaritifera*, for 2010, 2011 and January-March 2012 (n=960) showed a mean concentration of 9 mg/l with a range of 5-137 mg/l. The lower Blackwater had a mean value of 16 mg/l, Nore 8 mg/l, Slaney 7 mg/l and Aherlow 6 mg/l which are all well below the guideline threshold of 25 mg/l in the Freshwater Fish Directive and the standard in the Irish Salmonid Regulations. All four rivers (n=857) showed a statistically significant correlation between colour and suspended solids and an inverse relationship between colour and conductivity while the Aherlow and Nore also exhibited an inverse association between conductivity and suspended solids. The conclusion from the examination of data-sets of easily measured parameters, as surrogates for suspended solids, is that these can vary between rivers and that each site is different and thus can yield different relationships. Any relationship developed between surrogate parameters of suspended solids would be largely nullified by the amount of work required gaining adequate data over a range of hydrographical conditions.

In contrast with suspended solids levels, the nitrite limit value (0.05 mg/l NO_2), in the Irish Salmonid Regulations, is exceeded in south-eastern and other Irish salmonid-designated rivers on a recurring basis. Nitrite and nitrate results for designated salmonid rivers in the south-east, which also support *M. margaritifera*, for 2010, 2011 and January-March 2012 (n=1238) showed mean concentrations of 0.011 mg/l NO_2 and 2.6 mg/l N with ranges of <0.002-0.058 NO_2 and 0.05-5.84 mg/l N. The lower Blackwater had mean values of 0.010 mg/l NO_2 and 2.9 mg/l N, Nore 0.012 mg/l NO_2 and 2.3 mg/l N, Slaney 0.011 mg/l NO_2 and 3.4 mg/l N and Aherlow 0.010 mg/l NO_2 and 2.1 mg/l N which are all at the guideline threshold of 0.01 mg/l NO_2 in the parent Freshwater Fish Directive and would exceed the <1.7 mg/l N value suggested for protection of Irish pearl mussels. A sequential increase in nitrate levels with passage downstream is apparent in the Nore and the other rivers. Of 31 river study sites, for which data were available, only one in the Nore, Mountain, Multeen and Thonoge would be regarded as satisfactory in 2010 for pearl-mussel populations in respect of nitrite+nitrate-concentration conditions of <1.7 mg/l N. The mean nitrate value associated with the highest ecological quality (Q5) in Irish rivers is more than one-half lower again at 0.76 mg/l N and only one site would achieve that quality. With regard to the other nutrient phosphate, the mean value for the four salmonid-designated rivers was collectively and individually 0.04 mg/l P which is above the water quality criterion for high status (≤0.025 mg/l P) in the Regulations implementing the Water Framework Directive (WFD).

Dissolved oxygen studies, of water and river sediments, were carried out at 56 sites on 14 rivers overall. Redox potential readings were also taken at most of the sites for comparison. While no overall correlation was found between oxygen and redox readings there was a statistically significant relationship between the parameters in the two Barrow tributaries – the Mountain and Ballymurphy rivers. Generally the redox potential was higher when water conductivity was lower. However, although a statistically significant inverse relationship (P=0.0042) was found for redox and conductivity in the hard-water Nore river this did not hold true when water results for all river sites were computed (P=0.0894). During the laboratory redox measurements trials it was found that results were very variable, e.g. rise in gravel readings from 181 to 250 mV in two-minute period and still rising reaching 265 mV after 10 minutes. Even in deoxygenated conditions the same trend was evident. For all redox measurements for the project the initial reading, on insertion of probe, was used as there was significant drift with time. Redox potential measurements were more easily made in the laboratory trials than in the field. The practical use of redox potential measurements in natural ecosystems is fraught with difficulty and results should really be used in relation to other environmental parameters. With regard to oxygen the redox potential should be regarded more as a relative rather than an exact measurement.

Some indications of silt-induced interstitial deoxygenation were found in a few of the rivers studied. The substrata in the River Nore at locations respectively 38 and 68 km from source, during a four-week period, showed no significant difference in mean oxygen saturation levels with overlying water column. Similarly, the River Suir at Marlfield showed 99, 94 and 94 per cent respectively in water column, 3cm and 10cm sediment depth. On the other hand three sites in its tributary the Multeen, which were silted but have relatively low nitrite and nitrate levels, had average oxygen readings of 104, 69 and 57 per cent saturation for surface water and the same sediment depths. Overall in riffle/glide areas of rivers tested, average oxygen values were not significantly different in water and substrata.

The most important outcome of the study is that a reliable method of directly measuring dissolved oxygen in river substrata has been tested in laboratory and field conditions. Redox potential readings were found to be very variable under field conditions and no definitive conclusion can be made regarding its reliability as a water/sediment quality parameter. Hitherto, direct measurement of dissolved oxygen in river substrata has not been reported. Given the instrumentation available, it is surprising that the present study appears to be the first time that the direct measurement of dissolved oxygen has been carried out in river substrata. The advent of optical methods has made it possible to directly measure oxygen in such situations.

Only in gravel/sand substrata was penetration by the oxygen probe possible to 10cm or beyond. In most rivers in riffle and glide areas a hardpan was encountered preventing penetration to deeper depths in substratum. The oxygen probe would benefit by being slimmer for penetration into such substrata. Compared with the redox electrode it did allow more ingress of water while being inserted to take measurements although settlement time was allowed in the case of oxygen measurements whereas redox results were taken immediately on insertion. With both techniques as much care as is possible was taken not to introduce air or oxygenated water into the sediment to be measured.

Among the habitat needs for the freshwater pearl mussel, the number of juvenile salmonids has been given as ≥5 per 100 m^2, or 0.05 per m^2, for summer surveys. Data from Inland Fisheries Ireland's rapid fish surveys indicate that 10 sites in the study, for which data were available, would fulfil the *M. margaritifera* requirement in relation to salmonid presence. A site in the Nore, albeit slow flowing, with estimates of 0.001 and 0.01 per m^2 respectively for salmon (*Salmo salar*) and trout (*Salmo trutta*) would not apparently fulfil this criterion while one in the Multeen indicated a presence of 0.05 and 0.04 per m^2 for these fish. Because the salmon, in freshwater, is also protected under the EU Habitats Directive it would make sense for the two species' conservation, i.e. pearl mussel and salmon, to be considered together, in an integrated mode, rather than in isolation.

Overall the availability of salmonid host-fish would appear not to be a limiting factor for the mussel's life-cycle but both are affected directly and indirectly by nutrient enrichment and siltation effects.

Visual observations of siltation, at the 56 sites covered in the study, indicated that 35 were clean, 16 were clean with some degree of siltation at margins, three were slightly silted and two others were slightly to moderately affected. It would appear that while some siltation is evident at some river sites, no indications were found of widespread silt deposition. Further, there is no evidence, from an examination of data, that suspended solids in four salmonid/*Margaritifera* rivers in the south-east are excessive. Whatever about siltation, and its associated effect on oxygen, it would appear that nitrate concentrations, which have increased in Irish rivers over the past 30 years, may be implicated in the decline of pearl-mussel populations. *Ipso facto*, it can be tentatively concluded that, despite siltation being a problem at some sites, current nitrate levels in south-east Irish rivers would appear to be a more widespread risk for pearl-mussel survival.

Soil erosion can result in nutrient and sediment loss to waterways. Depending on rainfall the suspended solids loss from grazed grassland in Ireland can range between 45 and 90 kg per hectare which is relatively low compared to losses of several tonnes per hectare that can occur with severe soil erosion under tillage conditions. Any remedial measures to improve the mussels' and fishes' habitat, if focused on reducing fine material transport into streams, would also help to reduce diffusion of nutrients.

INTRODUCTION

This report is based on a study carried out during 2010 and 2011 as part of a FÁS Graduate Work Placement Programme Scheme at the Kilkenny Regional Laboratory of the Environmental Protection Agency (EPA).

BACKGROUND

The input of excessive quantities of sediment into salmonid spawning beds (redds) and juvenile pearl mussel (*Margaritifera margaritifera*) habitats has been identified as representing a key limiting factor to both populations' successful reproduction (e.g. Chapman, 1988; Buddensiek *et al.*, 1993; Heywood and Walling, 2007; Moorkens, 2010).

Sediment can arise from a possible number of sources including:

- Draining/dredging/realigning or other morphological effects on the watercourse;
- Cultivation of land adjacent to the watercourse;
- Run-off from agricultural areas, particularly tillage, close to the watercourse;
- Quarrying or other extractive activities close to the watercourse;
- Cattle poaching and erosion of river-banks;
- Forestry planting and clear-felling;
- Changes to flow, e.g. due to impoundment or abstraction of river water.

As well as having direct and indirect effects on the river fauna, sediment can also carry nutrients and other pollutants, such as pesticides, metals and faecal bacteria, into the watercourse. The accumulation of sediment on and within redd gravels can have lethal and sub-lethal effects on the salmonid eggs and sac fry that incubate within the gravel voids. While it is believed that fine sediment can smother embryos and prevent emergence of fry from the gravel bed, the principal mechanism whereby redd sedimentation adversely affects embryo survival involves the reduction of dissolved oxygen to the eggs (Heywood & Walling, 2007). Apparently, no direct *in-situ* methods have been used to measure dissolved oxygen in salmonid redds or pearl-mussel beds. Usually intragravel water is extracted for subsequent analysis (e.g. Ingendahl, 2001; Heywood & Walling, 2007). While no direct measurements of dissolved oxygen in substrata occupied by juvenile pearl mussels have, evidently, been made (e.g. Skinner *et al.*, 2003), the levels tolerated have been measured for four rivers in Germany (Buddensiek *et al.*, 1993). The relative extent of sediment oxidation can, it is claimed, be measured by redox potential (Aldridge and Ganf, 2003) and this parameter, which bears more relation to conductivity and pH than oxygen, has been used as a surrogate in recent pearl mussel studies (e.g. Geist & Auerswald, 2007; North South 2 Project, 2009).

Substratum characteristics are likely to be the main factors determining pearl mussel abundance in different parts of a river as has been found in a Scottish study (Hastie *et al.*, 2000). While some fine interstitial sediments are necessary for burrowing by juvenile mussels, artificial siltation of their habitat will prevent exchange of oxygen with the water column.

OBJECTIVES AND AIMS

The main objectives of the project were:

- To examine silt accumulation along the River Nore;
- To examine relationship between turbidity and suspended solids;
- To investigate whether dissolved oxygen could be measured successfully in interstitial water using a luminescent dissolved oxygen probe;
- To explore relationships between dissolved oxygen and redox potential in interstitial water.

Sedimentation in rivers is difficult to measure but sediment deposition estimates in a river can be quantified with sediment traps placed in the substratum. While sediment-trap measurements are only representative for the site and time of exposure, the data can be used to calculate sedimentation rates of river sections (Kozerski, 2002). Sedimentation or the accumulation of fines causes a decrease in the availability of dissolved oxygen in the river bed to juvenile mussels as well as salmonid eggs and alevins which inhabit the river bed during this part of their lifecycle. These conditions do not allow for the successful recruitment of juvenile mussels (Moorkens, 2010). Previous research (Geist & Auerswald 2007) and the Irish *Margaritifera* Management Plans, describe how measurements of dissolved oxygen in the river bed were not carried out because reliable instrumentation was not available, instead redox potential was used as a 'proxy' for measuring this parameter (North South 2 Project, 2009). In the project reported on here, dissolved oxygen was measured directly in the interstitial river water using an optical dissolved oxygen meter.

DISSOLVED OXYGEN

Oxygen is the most abundant element in the earth's crust (49.2% by weight) and is present in the atmosphere (28% by volume). Atmospheric oxygen is of vital importance for all organisms that carry out aerobic respiration. Originally dissolved oxygen (DO) in water was determined by the procedure devised by Winkler (1888) where precipitation of manganous hydroxide is brought about in a glass-stoppered bottle completely filled with the sample under test. Any oxygen present in solution then quickly combines with the manganous hydroxide forming higher hydroxides which, on subsequent acidification in the presence of iodine, liberate iodine in an amount chemically equivalent to the original dissolved oxygen content of the sample. The iodine is then determined by titration with a standard solution of sodium thiosulphate and the result expressed in mg/l.

$$\text{Dissolved oxygen content} = \frac{\text{Volume N/80 thiosulphate (ml) x 101.6}}{\text{Volume titrated (ml)}} \quad \text{mg/l}$$

Temperature was usually also recorded and the result could then be represented as per cent saturation by reference to an interpolation table. This method relied on the water sample being 'fixed' at the site by addition of chemical reagents prior to subsequent analysis in the laboratory. This procedure has now largely been superseded by direct measurement in the field by instrumental means which give readings in both mg/l and per cent saturation. There are three main types of instrument for determining dissolved oxygen in water: meters with a polarographic sensor, a galvanic sensing probe and the newer type with the optical fluorescence sensor. The optical type dissolved oxygen probe does not require water movement or stirring for measurement and was chosen after testing for the current project. Dissolved oxygen concentration is probably the most important variable related to the overall well-being of aquatic ecosystems.

Sources: Winkler, 1888; Department of the Environment, 1972; Oxford Reference Shelf, 1994; Boyd, 2000; EPA, 2001.

REDOX POTENTIAL

Oxidation–reduction (redox) was originally simply regarded as a chemical reaction with oxygen. The reverse process – loss of oxygen – was called reduction. Reaction with hydrogen also came to be regarded as reduction. Later, a more general idea of oxidation and reduction was developed in which oxidation was loss of electrons and reduction was gain of electrons. In aqueous solutions, the reduction potential is a measure of the tendency of the solution to either gain or lose electrons when it is subject to change by introduction of a new species. Like pH, the reduction potential represents an intensity factor. It does not characterize the capacity of the system for oxidation or reduction, in much the same way that pH does not characterize the acidity.

The main value of the redox potential is for explaining how oxidations and reductions occur in sediment-water systems. Oxidation-reduction potential can be either calculated or measured with an instrument. The instrument used in the present study consisted of a platinum electrode and a reference electrode connected to a pH meter. Redox potential is measured as the voltage between the platinum and reference electrodes with the measured redox potential then corrected for temperature. Redox measurements (Eh) are expressed in mV (millivolts). The reading is a voltage, relative to the reference electrode, with positive values (e.g. +300 mV) indicating an oxidizing environment (ability to accept electrons) and negative values (e.g. – 300 mV) indicating a reducing environment (ability to furnish electrons). Water samples cannot be preserved and stored for the Eh measurements and these must be carried out *in situ*. While measurement of Eh in water is relatively straightforward many factors limit the interpretation of these values. It is not regarded, by some, as a useful variable in water quality criteria for aquatic ecosystems.

Sources: Oxford Reference Shelf, 1994; Nordstrom & Wilde, 1998; Boyd, 2000; American Public Health Association *et al*., 2005.

NATURAL AND ARTIFICIAL SILTATION

River flows, especially high flows, continually move and sort river sediments into features such as pools, glides, riffles, bars and islands creating a range of habitats important for plants and animals. Water naturally carries sediment downstream gradually over time, particularly during high flows and floods. Water also naturally erodes sediment from the bed and banks of rivers and transports it downstream through the catchment, depositing it in areas where the flow is slower and land flatter.

Rare flood events with very high flows can transport more sediment than more frequent lower flow flood events, leading to a sudden accumulation of sediment. Other events such as landslides can also result in an oversupply to the river system, leading to a sudden accumulation of sediment; a fish kill occurred in two rivers in north Kerry (Glashoreag and Smearlagh) in August 2008 as a result of a bog-slide in which an estimated 5,000 fish perished, due to smothering effects, and in 2003 a similar peat-slide event occurred at Derrybrien, in county Galway, which also resulted in fish deaths. Many important species rely on river sediments for survival. For example, Atlantic salmon and lamprey migrate upstream as adults into rivers to reach spawning areas, which are normally clean stony or gravelly stretches of faster flowing water. They spawn in these areas, laying eggs in redds (nests) made by moving stones to form a shallow depression. After hatching, both species move to other parts of the river to feed and grow. Freshwater pearl mussels are globally very rare and Ireland supports some of the most important populations in the world for this species. Freshwater pearl mussels require clean river sediments and live buried in coarse sand or fine gravel and rely on a healthy salmon or trout population for the dispersal of juveniles.

For the purposes of aquatic monitoring sediment can be classified as deposited or suspended. Deposited sediment is that found on the bed of a river while suspened sediment (or suspended solids) is that found in the water column where it is being transported by water movement.

The deposition of the fine particles on the surface or within the stream bed is referred to as siltation. When the rate of deposition is unnaturally high, mainly due to human activities, problems arise primarily with the smothering of coarse patches of sediment with fine particles that ingress into the coarse sediment and deplete oxygen levels by reducing through-flow within the sediment. The settling of particles under low flow conditions can cause what is knoiw as external colmation – colmation is the retention processes that can lead to the clogging of the top layer of channel sediments.

Increased fine sediment deposition in rivers can arise from a combination of factors including low flows, as was evident in the summer of 2011, habitat modification and excessive sediment delivery from the catchment. The intensification of arable and livestock farming during the late 20th century has led to artificial siltation. While the soil particles themselves can threaten freshwater pearl mussel habitat, agricultural soils are also rich in nutrients and soil erosion is one of the ways by which phosphorus is transported from the terrestrial to aquatic environment. Phosphorus is a major contributor to freshwater eutrophication. Plantation forestry may also contribute to siltation and is another important land-use in many of the catchments that support freshwater pearl mussels. Some of the coniferous plantations of the late 20th century have trees that are planted up to the stream edge and drains which may connect directly to stream channels.

Over the last century sediment transport has increased by a factor of 100 in some of the rivers of northern Europe. Many rivers carry increased amounts of fine sediment as a consequence of intensive land use and high surface runoffs, which often leads to reduced oxygen supply in the interstitial zone.

In 2009 an international seminar was held at Clervaux in Luxembourg to discuss the problems that fine sediments have on rivers and their biota with special emphasis on freshwater mussels.

Sources: Ongley, 1996; Brunke, 1999; Langan *et al.*, 2007; Hall, 2008; Lucey, 2009; Project LIFE-Nature, 2009; SEPA (Scottish Environment Protection Agency), 2010; Extence *et al.*, 2011.

Figure 1 Showing localized siltation, from cattle poaching, in the River Licky (Co Waterford) and erosion of river bank in Camcor River (Co Offaly)

MATERIALS AND METHODS

Dissolved oxygen measurements were carried out using a hand-held meter, with luminescence technology (YSI ProODO) which eliminates sensor flow dependence, fitted to a probe with a stainless steel guard. To further protect the sensor and guard a 63µm nylon mesh (EFE & GB Nets) 'sock' was fitted to the probe and used in both water and sediment interstitial measurements. The probe was inserted into the river substratum at an angle of approximately 45° at depths of 3cm and 10cm, where possible, and readings taken after a three-minute stabilization period. Dissolved oxygen readings are expressed as per cent saturation (%).

Figure 2 YSI Luminescence Oxygen Meter

Redox potential measurements were carried out using a WTW modified pH instrument (ELANA Boden-Wasser-Monitoring). This consisted of a platinum electrode and a reference electrode, silver/silver chloride (Ag/AgCl) with potassium chloride (KCl) electrolyte, connected to a pH meter. Redox potential was measured as the voltage between the platinum and reference electrodes with the measured redox potential not corrected for temperature except where stated in the text. For water measurements both electrodes were suspended in the water column while for sediment measurements the platinum electrode was inserted to depths of 3, 5 or 10cm, where possible, and readings taken immediately (*Instruction manual for the SCHOTT® Instruments electrodes for measurement of pH values and redox potential*). Because of variation between initial and delayed readings it is recommended that readings are taken immediately (North South 2 Project, 2009). Redox potential readings are expressed in mV (millivolts).

Figure 3 WTW Redox Potential Meter

In the laboratory testing of dissolved oxygen and redox meters, for reproducibility and repeatability, tap water and clean aquarium gravels were used. To deoxygenate the water and interstitial water, 20g of sodium sulphite (Na_2SO_3) were dissolved in the one-litre test beaker and left for 60 minutes to attain a dissolved-oxygen free solution (YSI, 2009).

At two locations on the River Nore, Turbidity was measured *in situ* using YSI 6600 Water Quality Sonde data loggers while suspended solids were determined in the laboratory. Samples for suspended solids were collected using one-litre plastic bottles which were rinsed out several times with the river water before final collection of the water sample. Samples were stored in a refrigerator at 3-5˚C and analyzed, within seven days, by filtering 500 ml of sample through 47mm Whatman 934-AH Glass Microfibre filters and drying at 105˚C ± 2.0˚C. The data loggers measured turbidity at fifteen-minute intervals over the four-week period that the sediment traps were *in-situ*. Turbidity is expressed as Nephelometric Turbidity Units (NTU)[1] and suspended solids in mg/l.

1 There is a relative multiplicity of units for turbidity in water but apart from those expressed in silica units the three other main units used, i.e. FTU (Formazin), JTU (Jackson) and NTU (Nephelometric), are virtually equivalent and are used interchangeably (EPA, 2001).

Figure 4 Sonde Water Quality Data Logger

Sediment traps were used to measure siltation accumulation at two locations on the River Nore (Figure 5). These were 2.5 litre buckets with an internal diameter of 180mm and an internal depth of 135mm containing sieved particles ranging in size from 2 to 19mm. The material used for the sediment traps was sourced from the River Nore at Kilbricken one of the sites used in the study. The buckets were buried in the river, flush with the top of the river bed and lids were removed upon deployment. The lids were replaced on the buckets for collection to eliminate any loss of fines (Hall, 2008). The traps were placed in a line along the river with a second line added to act as a replicate. A set (2) of traps was removed every seven days over the four week period of the experiment from each site.

The sedimentation rate is reported as grams per square metre per day (g/m^2/d) and calculated as follows:

= total dry weight of sediment collected from collecting buckets, g/area of collecting buckets, m^2/total days of trap deployment

= dry weight, g/0.02544 m^2/day

Particle Size Analysis was used to determine the particle size distribution of sediments. The main focus was on the fine material <2mm, which clogs the interstitial zone and decreases the amount of oxygen available to the benthic inhabitants. Grain sizes were fractioned with a wet sieving tower of decreasing mesh sizes (2mm, 500µm, 250µm, 125µm, 63µm). The wash water, containing the 63µm fraction was collected and allowed to settle for a period of five days when the water was siphoned off. The fractions retained on each sieve were dried at 105˚C and left in a glass vacuum desiccator for 1.5 hours before being weighed.

Figure 5 Setting Sediment Traps in River Nore

River flow (m^3/S) was also measured at each of the sites over the four week period, using an OTT current meter (OTT C20) with wading rod (Ott Messtechnik GmbH & Co. KG), by the velocity-area method.

Battery pack electric-fishing for juvenile salmonids was carried out by Inland Fisheries Ireland (IFI) staff during catchment-wide electric-fishing (CWEF) surveys. CWEF is a semi-quantitative timed (5-minute) electric-fishing sampling method. The salmonid-fish data included for some of the study sites are reported as number of salmonids per m^2.

Finally, physico-chemical data (Neill, 2010; Bowden, 2011 and 2012) are expressed as follows: nitrate (nitrate+nitrite: TON) as mg/l N, phosphate (ortho-phosphate: MRP) as mg/l P, chloride as mg/l Cl and colour, either true (0.45µm filter) or apparent (unfiltered water), as Hazen[2]. The levels cited are generally mean values based on at least four samples taken at each site throughout the year.

2 From January 2011 samples for colour (Hazen) analysis at Kilkenny EPA were filtered using a standard 0.45µm filter (E. Holohan, personal communication).

STUDY AREAS

The field studies were carried out on pearl-mussel rivers in the south-east of Ireland with emphasis on the River Nore which is home to the last surviving population of the hard-water form (ecophenotype) of *Margaritifera margaritifera* (Lucey, 2006).

Figure 6 Showing pearl mussel with incrustation and lack of umbonal erosion from the hard-water River Nore (above) and juvenile pearl mussels with typical umbonal erosion from soft-water river (below)

The two sites on the River Nore used for the siltation accumulation phase of the study were at Kilbricken Bridge and Ballyragget Bridge, respectively 38 and 68 km from source, during a four-week period Table 1).

Table 1 Hydrological Characteristics of River Nore Sites at Kilbricken and Ballyragget

Location	Catchment Area (sq km)	Slope (%)	Distance from Source (km)	Stream Length (km)
Kilbricken	343.3	5.5	38	396.3
Ballyragget	1059	4.0	68	966.5

The sites could be classified as glide areas having a slope difference of 1.5 per cent in the 30 km between the two. For the siltation studies glide areas, of primarily gravel and sand, were used as such substrata represent juvenile pearl mussel and salmonid habitats (Á. O'Connor, personal communication).

Field work for dissolved oxygen studies was conducted during 2010 and 2011 when 56 locations on 14 rivers were visited on up to five occasions. The geographical area of the rivers examined was principally in the south-east of Ireland (See Figure 7). For the purposes of this study the rivers were divided into two groups. The first set of rivers (Table 2a), which encompassed most of the research are protected under the Freshwater Pearl Mussel Regulations (European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009. This group of rivers includes the River Nore which contains the ecophenotypic form of *M. margaritifera* aka *M. durrovensis*. This river was chosen to examine the accumulation of sediment over a four-week period and its relationship with turbidity, suspended solids and interstitial dissolved oxygen levels. Figure 7 shows the Group 1 SAC (Special Area of Conservation) rivers (CEC, 1992) examined in this study[3]. The second set of rivers are located in the same geographical area but are not protected under the Freshwater Pearl Mussel Regulations. These rivers are important in relation to *M. margaritifera* because populations of this bivalve have been recorded previously, by one of the authors, and some populations may remain. These rivers, with locations, are also listed in Table 2b.

3 There are currently 19 SACs, covering 27 sub-basins, designated for the pearl mussel throughout the Republic of Ireland.

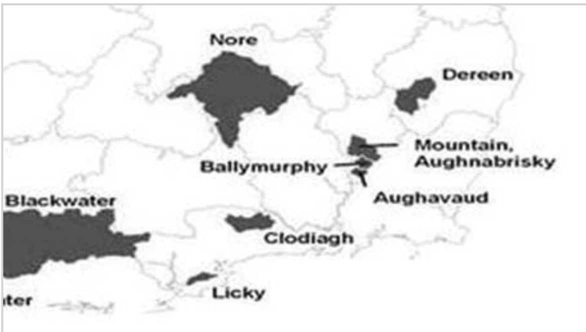


Figure 7 Map Showing Main Irish Rivers and Study Core River Catchments (Detail is from North South Project Map)

Table 2a List of rivers protected under the Freshwater Pearl Mussel Regulations, 2009 with locations sampled

River	Sampling Locations	SAC Site Name
Nore	Kilbricken Bridge Poorman's Bridge Waterloo Bridge Watercastle New Bridge Tallyho Bridge Ballyragget Knapton Bridge Threecastles Maddockstown Inchbeg Warrington	River Barrow and River Nore
Blackwater	Ballyduff Bridge Lismore Bridge 2 km d/s Lismore u/s Cappoquin	Blackwater River (Cork/Waterford)
Derreen (Slaney)	Ballykilmurray Knockeen Rathnagrew Rathglass Ballyduff	Slaney River Valley
Mountain (Barrow)	Upstream of Viaduct Kiledmond Rossdelig Owlbeg Lacken Bridge Borris u/s Barrow	River Barrow and River Nore
Ballymurphy (Barrow)	Ballyroughan Little Earl's Bridge Cullentragh	River Barrow and River Nore
Aughavaud (Barrow)	Turra Bridge St. Mullin's	River Barrow and River Nore
Clodiagh (Suir)	Clonea Bridge Glenstown Bridge Greens Bridge Lowry's Bridge Portlaw Bridge	Lower River Suir
Licky	Carrigeen Grallagh Licky Bridge	Blackwater River (Cork/Waterford)

Table 2b List of other rivers with locations sampled

River	Sampling Locations
Slaney	Aghade Bridge Rathmore Bridge Motaboher Bridge Kilcarry Bridge
Multeen	Black Bridge Aughnagross Ballygriffin
Aherlow	Old Cappa Bridge Kilardry Bridge
Suir	Marlfield
Tar	Kilganny Bridge Goat's Bridge Upstream Tar Bridge
Thonoge	Ballyboley Tubbrid

The general reference condition for Irish rivers is that they should have viable populations of salmonid fish (McGarrigle *et al*., 2010). Some of the rivers in the present study (Aherlow, Blackwater, Nore and Slaney) are designated salmonid waters in the context of the European Communities (Quality of Salmonid Waters) Regulations 1988. These Regulations implement the Freshwater Fish Directive (CEC, 1978) and specify a range of water quality parameters, including suspended solids (guideline annual mean value ≤25 mg/l), to be monitored. These four rivers also support pearl mussels and can be classed, based on alkalinity and hardness, as follows: Aherlow – slightly to moderately hard; Blackwater – slightly to moderately hard; Nore – hard; Slaney – slightly hard (Lucey, 2006).

RESULTS

SEDIMENT STUDIES

The sediment traps were deployed for a four-week period (6 August 2010 to 3 September 2010) at each of the two sampling sites in the River Nore (See Figure 8). The results of deposited fractions are displayed in Tables 3-5 and Figure 9.

The results of Particle Size Analysis are expressed in grams as mean values for the components of the sediment traps from the sites at Kilbricken and Ballyragget. The largest percentage of each sediment sample was sand. Flow, in (m^3/S), is shown in these tables also as an average over the corresponding week.

Both sites were glides and the flows during the study show relatively steady state conditions at the two locations. The fractions can be classified into groups as per the Udden-Wentworth scale (Udden, 1914; Wentworth, 1922), i.e. Fines (<63µm), Sand (63µm-2mm) and Gravel (2-4 mm), as in Tables 3 and 4. This classification shows a general increase of the three groups over the sample period, with the exception of the gravel fraction at the Ballyragget site.

The weight of the 2mm fraction for both sites was in the range of 103.32 to 154.9 grams (g). This fraction had the highest accumulation overall. The accumulation of the 500µm, 250µm, 125µm and 63µm fractions for both sites showed no trend in accumulation rates. The weight (g) of the fines (silt and clay) <63µm ranged from 1.3 to 1.82 between the two sites. There was an increase in the accumulated fines over week 1 to week 2, then a decrease in week 3 and an increase again. There was an increase of accumulation at the Kilbricken site from week 1 to week 3 and the weight of this fraction decreased for week 4. Using the results taken from each sediment trap and the flow rates from these sites during the sample period in which there was minimal rainfall, these data can be used as an indicator of baseline sediment flux.

The deposited fines (silt and clay), sand and gravel and other fractions are displayed, using log transformed data, in Figure 9. In Table 3 the mean levels (g) of accumulation of three sediment types from Kilbricken and Ballyragget over the period Week 1 to Week 4 are given with the average river flow for the week also shown. Table 4 shows the accumulation and sedimentation rate of all fractions for each week as well as totals. In Table 5 the mean rate of sedimentation (g/m^2/d) and average flow (m^3/S) for the two sites in the Nore are compared.

The mean sedimentation rate of fines at the two sites in the Nore, 8.73 g/m^2/d at Kilbricken and 8.83 g/m^2/d at Ballyragget, were very similar (Table 5). The same methodology of sediment trapping was used in the River Nore as had been employed in a study of the River Dawros in Co Galway (Hall, 2008; Hall & Allot, 2009). The sedimentation rate of fines at the two sites in the Nore ranged between 7.36 and 10.22 g/m^2/d, which is almost double the range recorded at the four sites in the Dawros, 4.18 to 5.87 g/m^2/d, during baseline flow conditions (Hall & Allott, 2009). During high flow in the Dawros, fines accumulated at a rate of up to 40 g/m^2/d (Hall & Allot, 2009).

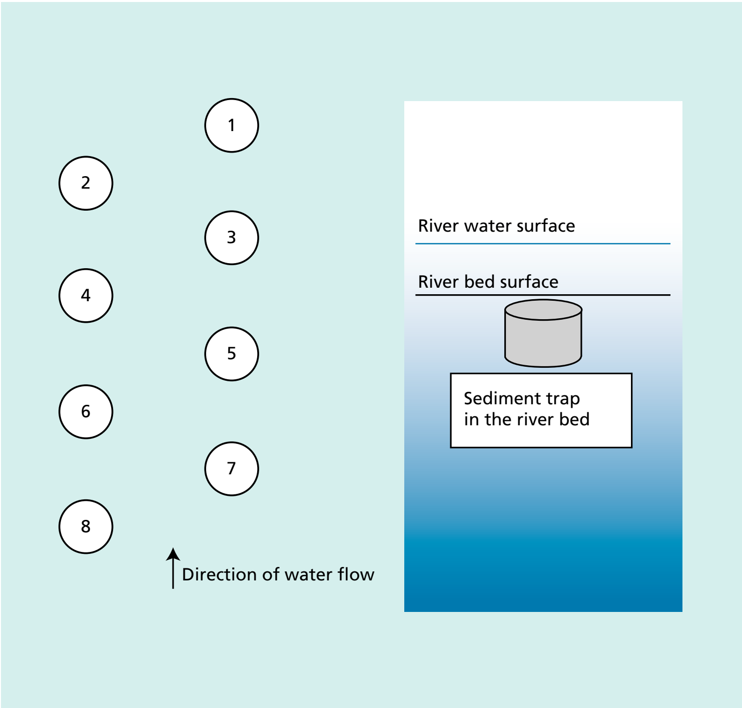


Figure 8 Showing sediment traps *in-situ* just downstream of Ballyragget Bridge in River Nore and schematic representation of position of the sediment traps with the direction of flow in the river indicated

Table 3 Particle size analysis results for sediment accumulation at Kilbricken (above) and Ballyragget (below). Results (grams) are shown for each particle size together with mean flow (m^3/S) for the corresponding week

	Week 1	Week 2	Week 3	Week 4
Particle Size				
Gravel	103.32	118.56	138.92	123.55
Sand	13.32	11.15	12.62	8.70
Fines	1.31	1.5	1.82	1.59
Flow (m^3/S)	0.78	0.62	0.65	0.60

	Week 1	Week 2	Week 3	Week 4
Particle Size				
Gravel	146.25	154.9	154.68	150.92
Sand	5.99	8.41	9.92	10.24
Fines	1.56	1.7	1.5	1.53
Flow (m^3/S)	3.15	3.07	3.58	3.00

In a segment of the lowland River Spree, in Germany, the effective average sedimentation rate ranged between 0.9 and 6.6 $g/m^2/d$ (Kozerski, 2003). In that study of the Spree, 10 km upstream of Berlin, using horizontal plate sediment traps, the variance of sedimentation was controlled by the suspended particulate matter concentration, the settling velocity of the particles and the flow velocity. The sinking velocity exhibited significant seasonal fluctuations in the Spree with highest values in summer. Obviously, the weight of the different factions will have different settling characteristics and will also not settle above certain flow velocities. The range in flow at Kilbricken and Ballyragget for the four-week period was respectively 0.16-1.1 and 2.4-4.0 m^3/S.

If the situation in the Dawros and Nore, where the same methodology was employed, are compared for stable base-flow conditions we see that there are some differences (Table 6) with the Nore exhibiting more accumulation generally except for sand at one site in the Dawros. In the Dawros, deposition was affected due to a humic layer forming on the surface of the contents of the trap which decreased the accumulation capacity of the traps over the extended period of time of five week deployment (Hall, 2008). In the Nore study the sediment traps were sampled weekly. The Dawros is described as a 'flashy' river in its upper reaches which is buffered by two lakes in its lower reaches (Hall, 2008) and the sites in the lower catchment would be more comparable to those in the Nore study.

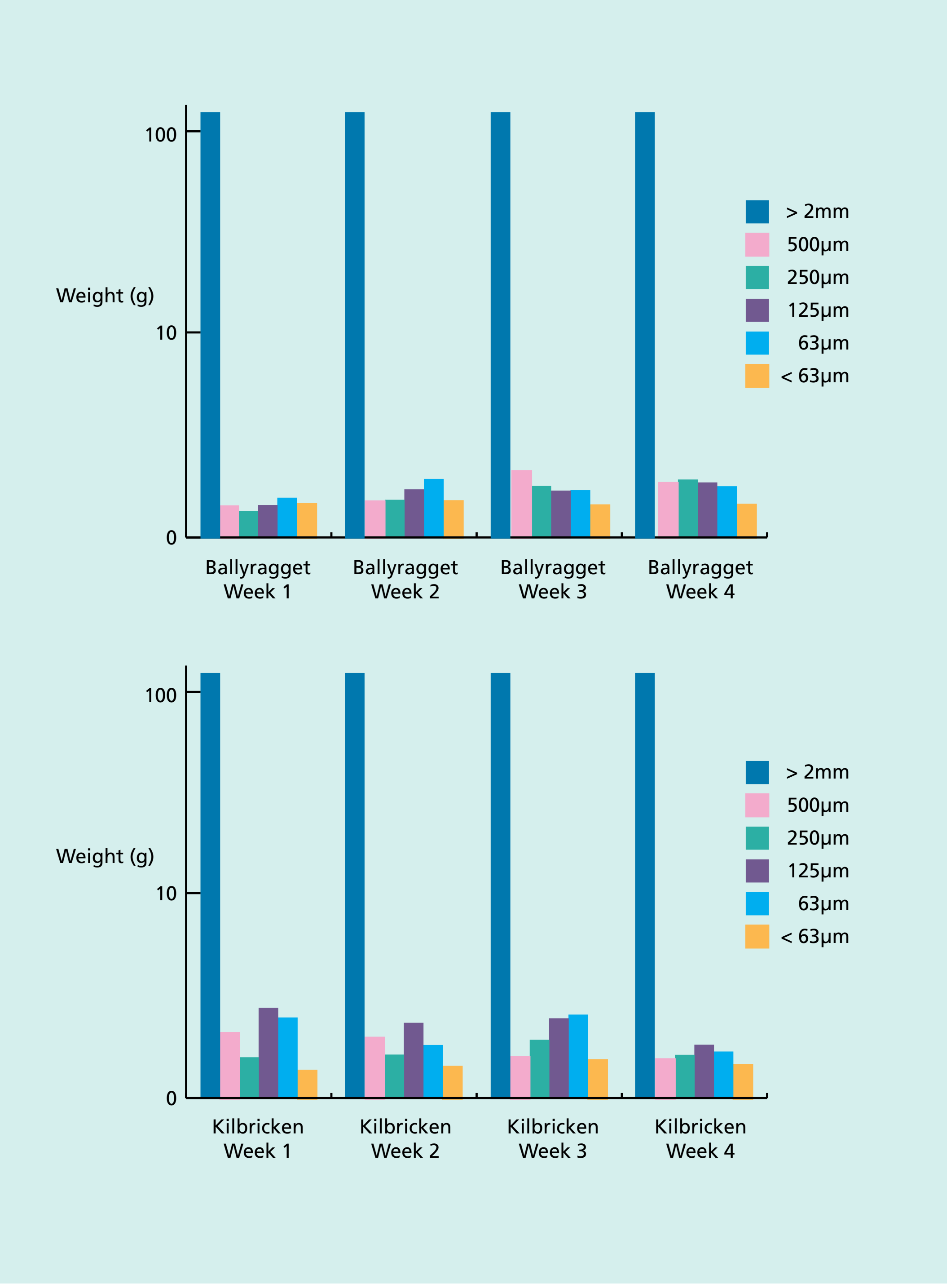


Figure 9 Showing the accumulated weight of different fractions, in grams $\log_{10}$, at Kilbricken (above) and Ballyragget (below) over the four-week sample period

Table 4 Accumulation and sedimentation rate of all fractions for each week and totals

Particle Size	Kilbricken Week 1	Kilbricken Week 2	Kilbricken Week 3	Kilbricken Week 4	Total	g/m²/d 28 days
> 2mm	103.32	118.56	138.92	123.55	484.35	680
500µm	3.15	2.92	1.97	1.87	9.91	13.9
250µm	1.94	2.07	2.79	2.06	8.86	12.4
125µm	4.35	3.62	3.84	2.55	14.36	20.2
63µm	3.88	2.54	4.02	2.22	12.66	17.8
<63µm	1.31	1.5	1.82	1.59	6.22	8.7
Total	117.95	131.21	153.36	133.84	536.36	753
g/m²/d	662.3	736.8	861.2	751.6	3011.9	753
Flow (m³/S)	0.78	0.62	0.65	0.60		

Particle Size	Ballyragget Week 1	Ballyragget Week 2	Ballyragget Week 3	Ballyragget Week 4	Total	g/m²/d 28 days
> 2mm	146.25	154.9	154.68	150.92	606.75	851.8
500µm	1.45	1.69	3.14	2.57	8.85	12.4
250µm	1.21	1.74	2.4	2.71	8.06	11.3
125µm	1.49	2.24	2.18	2.57	8.48	11.9
63µm	1.84	2.74	2.2	2.39	9.17	12.9
<63µm	1.56	1.7	1.5	1.53	6.29	8.8
Total	153.8	165.01	166.1	162.69	647.6	909.1
g/m²/d	863.7	926.6	932.7	913.6	3636.6	909.1
Flow (m³/S)	3.15	3.07	3.58	3.00		

Table 5 Mean rate of sedimentation and average flow for the two sites in the Nore

Area of collecting buckets = 0.02544 m^2 Mean Sedimentation Rate = 8.73 $g/m^2/d$

Location	Flow (m^3/S)	Days	Dry Weight of fines (<63μm) in collecting buckets (g)	Sedimentation Rate ($g/m^2/d$)
Kilbricken	0.67	7	1.31	7.36
	0.78	7	1.50	8.42
	0.65	7	1.82	10.22
	0.60	7	1.59	8.91

Area of collecting buckets = 0.02544 m^2 Mean Sedimentation Rate = 8.83 $g/m^2/d$

Location	Flow (m^3/S)	Days	Dry Weight of fines (<63μm) in collecting buckets (g)	Sedimentation Rate ($g/m^2/d$)
Ballyragget	3.15	7	1.56	8.76
	3.07	7	1.70	9.55
	3.58	7	1.50	8.42
	3.00	7	1.53	8.59

Table 6 Comparison of range in sedimentation rates ($g/m^2/d$) at sites in Dawros (Hall, 2008; Hall and Allott, 2009) and Nore rivers

	River Dawros	River Nore
Fines (<0.063 mm)	4.18-5.87	7.36-10.22
Sand (0.063-2 mm)	7.86-295.51	33.64-74.80
Gravel (2-4 mm)	0-143.51	103.32-154.90

SUSPENDED SOLIDS AND TURBIDITY

A preliminary set of measurements of turbidity (NTU) and suspended solids concentration (SSC) was carried out at Kilbricken, from 21-28 July 2010, when flows were also estimated (Figure 10). This showed a highly positive correlation between turbidity and suspended solids (r^2 = 0.9864) apparently corresponding with flow discharge; with high correlations for SSC v. flow (r^2 = 0.9696) and NTU v. flow (r^2 = 0.9882).

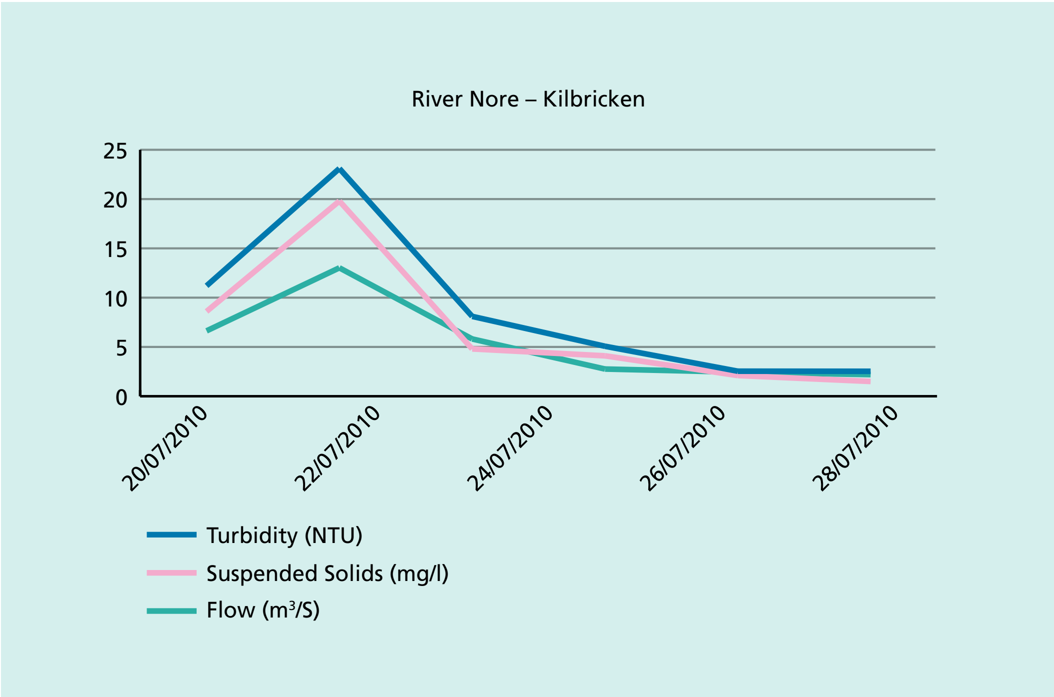


Figure 10 Showing preliminary measurements of turbidity (NTU) and suspended solids (mg/l) at Kilbricken with estimated river flow (m^3/S) in July 2010.

This would, prima facie, indicate that the NTU-SSC relationship should provide for the accurate estimation of SSC from turbidity measurements. However, in the study proper over a four-week period in lower flow conditions no such relationship was apparent. The results from the *in-situ* turbidity logger and the suspended solids measurements in August-September 2010 (Figure 11) show a poor correlation for both sites despite the parameters showing a similar graphical pattern over the period: Kilbricken (r^2 = 0.1873) and Ballyragget (r^2 = 0.0862) and with flow discharge with lower correlations at Kilbricken than Ballyragget for SSC v. flow (r^2 = 0.2295 and 0.5212) and NTU v. flow (r^2 = 0.0135 and 0.3240). Log transforming data did not improve the relationships significantly. The closest of these poor relationship between the two 'variables' is evident at Kilbricken.

From these results there is not a statistically significant relationship between turbidity and suspended solids at Kibricken: r = 0.4328 (P=0.0827), New Bridge: r = 0.1164 (P=0.6564) and Ballyragget: r = 0.2935 (P=0.2528).

Figure 11 Showing measurements of turbidity (NTU) and suspended solids (mg/l) at Kilbricken (above) and Ballyragget (below) during period of relatively stable river flow (m^3/S)

At an intermediate site, New Bridge, which is 20 km downstream of Kilbricken and 10 km upstream of Ballyragget, turbidity and suspended solid were also measured and the results for that location are shown in Figure 12.

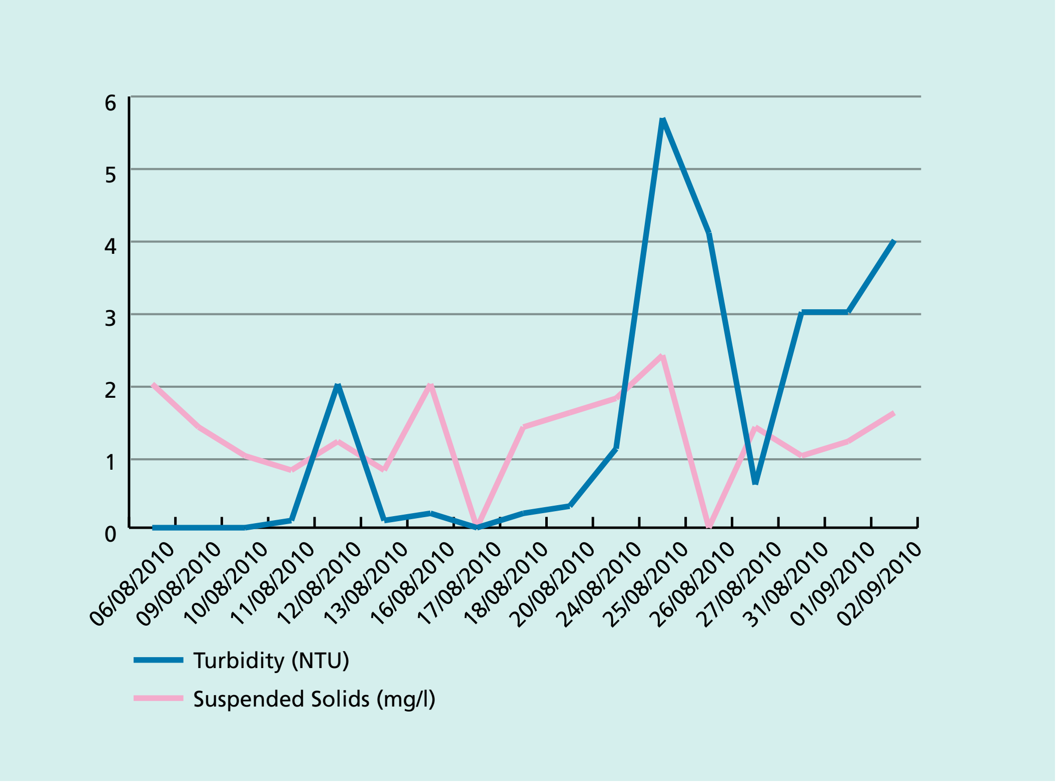


Figure 12 Showing measurements of turbidity (NTU) and suspended solids (mg/l) at New Bridge during period of relatively stable river flow

It can be seen that while there were some small differences in results for suspended solids, between the three sites, the mean values were practically the same (Table 7). For turbidity, however, there is a marked reduction in mean values going downstream which could indicate the contribution water colour was making to measurements at Kilbricken (Table 7).

Base-10 logarithmic transformation is one of several mathematical functions that can be used to transform data sets in order that the assumptions for linear regression analysis are met. Log transforming all data (Figure 13) did not significantly improve statistical relationship (P=0.3988) between turbidity and suspended solids (r^2 = 0.0146 and 0.0721).

Rainfall just before, during and after the study period was low (mean = 1.1mm) with a peak of 14.1mm on 22 August (Figure 14). That increase in precipitation is seemingly reflected in the turbidity and suspended solids values after the event (Figures 11 and 12). Not surprisingly no statistically significant relationship was found between rainfall and flow, turbidity or suspended solids with such a hysteresis effect, i.e. where changes in an effect lag behind changes in its cause.

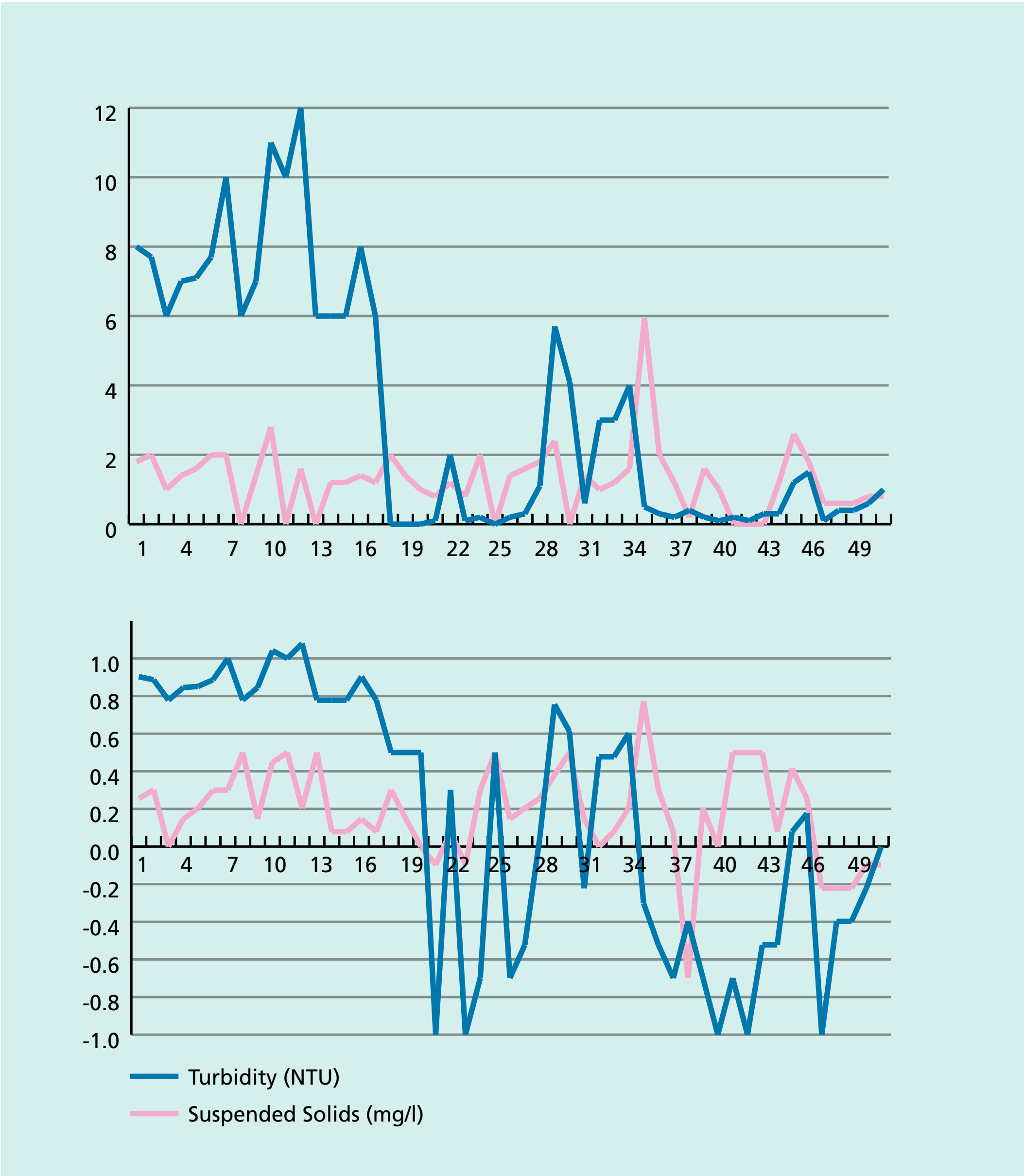


Figure 13 Showing measurements of turbidity (NTU) and suspended solids (mg/l) at all three sites (above) and with data log transformed (below)

Table 7 Range and means (in parentheses) for suspended solids and turbidity at three locations in River Nore

Location	Suspended Solids (mg/l)	Turbidity (NTU)
Kilbricken	0-2.8 (1.3)	6-12 (7.7)
New Bridge	0-2.4 (1.3)	0-5.7 (1.4)
Ballyragget	0-5.9 (1.2)	0.1-1.5 (0.5)

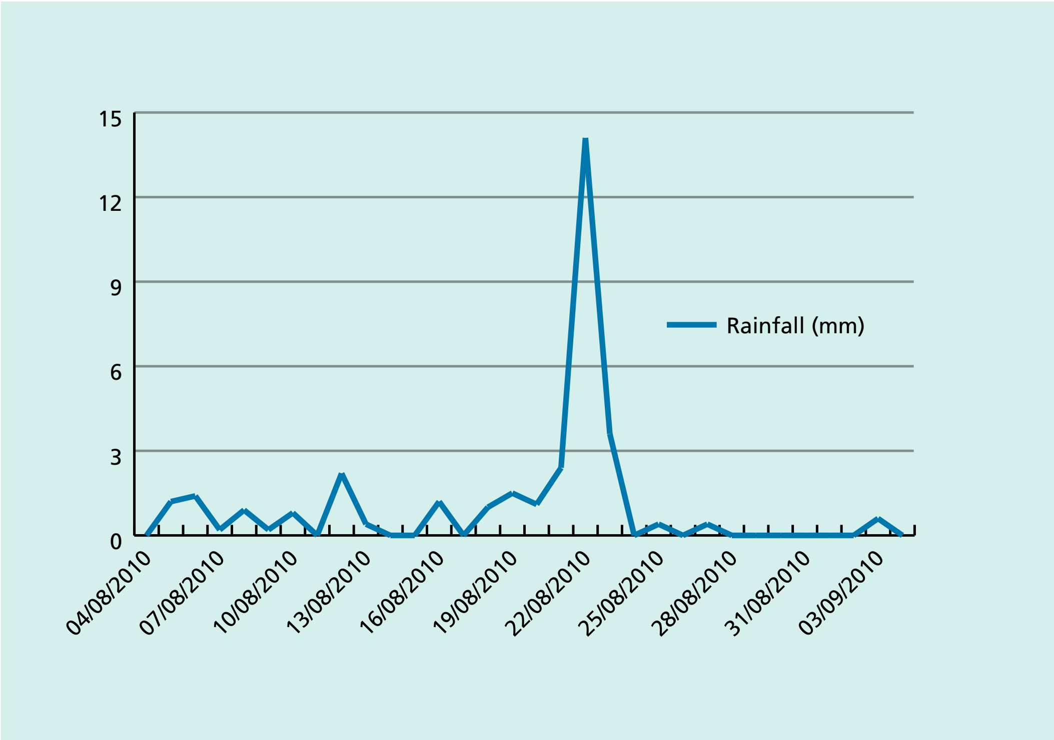


Figure 14 Rainfall (mm) near Cullahill Co Laois (from data supplied by Met Éireann)

Colour (Hazen) on the 5 August, the day before measurements started, was 167 at Kilbricken and 33 at Ballyragget site with annual means respectively of 75 and 35 for 2010. At New Bridge the corresponding values were 40 and 44. This colour difference reflects the presence of humic substances in the upper reaches where the river and its tributaries traverse areas of peat[4]. The upstream catchment characteristics comprise 15 per cent forestry and 10 per cent bogs at Kilbricken while the corresponding figures are halved at Ballyragget.

In 2010 the ecological water quality was Good (Q4) at both locations. Physico-chemical parameters (Bowden, 2011) also show relatively good conditions but with average nutrient and chloride values at the Kilbricken site approximately one-half of those at Ballyragget (Table 8).

4 The higher humic content of some Irish rivers is recognized in the the annex to the European Commission Decision on the Water Framework Directive (WFD) intercalibration results for Northern River Type (R-N1) which has derogation to < 150 mg/l Pt for organic material for Ireland from < 30 mg/l Pt for the other member states (European Commission, 2008). One Hazen unit is equivalent to 1 mg/l Pt.

Table 8 Mean values for some physico-chemical parameters at two locations in River Nore (from data in Bowden, 2011)

Location	Chloride (mg/l Cl)	Conductivity (µS/cm)	Nitrate (mg/l N)	Phosphate (mg/l P)
Kilbricken	13	440	1.2	0.02
Ballyragget	25	648	2.9	0.04

It would appear from the present study that turbidity values can be strongly influenced by water colour at some locations. The study yielded no clear relationship between turbidity and suspended solids with an r^2 value of 0.0146 (0.0720 $\log_{10}$ – transformed data) for all samples.

Conductivity is easily measured in the field or laboratory. Interestingly, this parameter showed an inverse relationship with colour. The colour and conductivity data for 2010 (Bowden, 2011) and 2011 (Bowden, 2012) for the three sites are plotted in Figure 15.

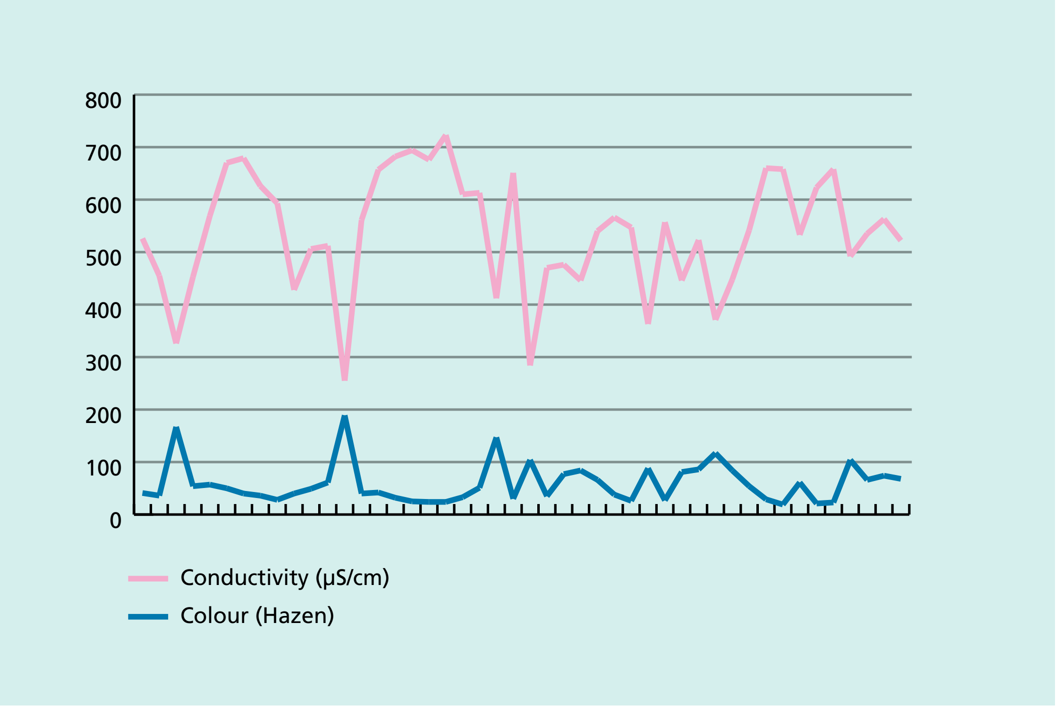


Figure 15 Showing measurements of colour (Hazen) and conductivity (µS/cm) at Kilbricken, New Bridge and Ballyragget in 2010 and 2011

In 2010, despite a high absolute linear correlation coefficient (r) there is not quite a statistically significant relationship between colour and conductivity at Kilbricken: r = – 0.9197 (P=0.0803), but only four sample results were available for that location. However, there is a statistically significant inverse relationship between these two parameters at New Bridge: r = – 0.8008 (P=0.0095) and Ballyragget: r = – 0.9574 (P=0.0001). When 2011 data are included, the relationship is significant at the three sites: Kilbricken r = – 0.7890 (P=0.0349): New Bridge r = – 0.8045 (P<0.0001); Ballyragget r = – 0.9066 (P<0.0001). Thus, an inverse relationship between colour and conductivity is indicated in the River Nore which becomes stronger with passage downstream from Kilbricken to Ballyragget.

Using data (2010, 2011 and January-March 2012) from the four salmonid – designated rivers in the south-east (C. Bowden, personal communication) the relationships between suspended solids, conductivity and colour were examined. SSC and conductivity (n=944) showed poor correlation (r^2=0.0148 and r^2=0.0247 log_{10} – transformed data); SSC and colour (n=921) showed poor correlation (r^2=0.0739 and r^2=0.1076 log_{10} – transformed data); conductivity and colour (n=974) showed poor correlation (r^2=0.0187 and r^2=0.0048 log_{10} – transformed data).

Table 9 Relationship between colour, conductivity and suspended solids in four salmonid-designated rivers in south-east Ireland

River	Parameters	Correlation Coefficient (r)	Number of Samples (n)	Significance (P)	Comment
Nore	Colour and Suspended Solids	0.3947	434	<0.0001	Extremely statistically significant
	Conductivity and Suspended Solids	-0.3728	434	<0.0001	Extremely statistically significant
	Colour and Conductivity	-0.6172	434	<0.0001	Extremely statistically significant
Aherlow	Colour and Suspended Solids	0.5346	103	<0.0001	Extremely statistically significant
	Conductivity and Suspended Solids	-0.3049	103	0.0017	Very statistically significant
	Colour and Conductivity	-0.6781	103	<0.0001	Extremely statistically significant
Slaney	Colour and Suspended Solids	0.1390	206	0.0463	Statistically significant
	Conductivity and Suspended Solids	-0.0512	206	0.4649	Not statistically significant
	Colour and Conductivity	-0.6974	206	<0.0001	Extremely statistically significant
Blackwater	Colour and Suspended Solids	0.4161	114	<0.0001	Extremely statistically significant
	Conductivity and Suspended Solids	0.0276	114	0.7707	Not statistically significant
	Colour and Conductivity	-0.5152	114	<0.0001	Extremely statistically significant

Table 10 Relationship between colour, conductivity and suspended solids at three locations in the salmonid-designated River Nore

River Location	Parameters	Correlation Coefficient (r)	Number of Samples (n)	Significance (P)	Comment
Upstream of Kilbricken*	Colour and Suspended Solids	0.5578	26	0.0031	Very statistically significant
	Conductivity and Suspended Solids	-0.5312	26	<0.0052	Very statistically significant
	Colour and Conductivity	-0.8023	26	<0.0001	Extremely statistically significant
New Bridge	Colour and Suspended Solids	0.2412	26	0.2352	Not statistically significant
	Conductivity and Suspended Solids	-0.2070	26	0.3103	Not statistically significant
	Colour and Conductivity	-0.8358	26	<0.0001	Extremely statistically significant
Ballyragget	Colour and Suspended Solids	0.5012	26	0.0091	Very statistically significant
	Conductivity and Suspended Solids	-0.6769	26	0.0001	Extremely statistically significant
	Colour and Conductivity	-0.8813	26	<0.0001	Extremely statistically significant

*This is 3 km u/s of Kilbricken

However, when the data for each river were treated separately they gave a statistically significant correlation between colour and suspended solids and an inverse relationship between colour and conductivity while the Aherlow and Nore also exhibited an inverse association between conductivity and suspended solids (Table 9).

The relationship between colour and conductivity was extremely statistically significant for the rivers and locations in Tables 9 and 10. As with the treatment of the lower number of samples above, for the three locations in the Nore, a strong inverse association between colour and conductivity was apparent which strengthened with passage downstream. Of the three locations only New Bridge did not indicate a statistically significant relationship for conductivity with suspended solids and colour with conductivity.

Table 11 Suspended solids measurements at three locations in the salmonid-designated River Nore in 2010 and 2011

River Location	Year	Range	Mean	Number of Samples (n)
Upstream of Kilbricken*	2010	5-30	8	10
	2011	5-18	8	12
New Bridge	2010	5-27	9	10
	2011	5-34	8	12
Ballyragget	2010	5-48	13	10
	2011	5-18	7	12

*This is 3 km u/s of Kilbricken

Table 12 Nitrite and nitrate measurements at three locations in the salmonid-designated River Nore in 2010 and 2011

River Location	Year	Range		Mean		Number of Samples (n)
		Nitrite (NO_2)	Nitrate (N)	Nitrite (NO_2)	Nitrate (N)	
Upstream of Kilbricken*	2010	0.001-0.013	0.6-1.6	0.006	1.1	10
	2011	0.002-0.012	0.1-2.1	0.005	1.1	10
New Bridge	2010	0.001-0.027	0.8-3.1	0.013	2.2	10
	2011	0.005-0.015	1.1-2.7	0.005	2.0	10
Ballyragget	2010	0.007-0.028	1.6-3.4	0.016	2.8	10
	2011	0.006-0.058	1.7-3.5	0.016	2.5	11

*This is 3 km u/s of Kilbricken

The three sites in the Nore would easily comply with the suspended solids standard in the Irish Salmonid Regulations and the Freshwater Fish Directive guide value which are set at 25 mg/l (Table 11).

NITRITE, NITRATE AND PHOSPHATE

In contrast with suspended solids levels, the nitrite limit value (0.05 mg/l NO_2), in the Irish Salmonid Regulations, is exceeded in salmonid-designated rivers on a recurring basis (Toner *et al*., 2005). The guide value in the parent Freshwater Fish Directive (0.01 mg/l NO_2) would be marginally exceeded at Ballyragget in 2010 and 2011 (Table 12). As with nitrate levels these show an increase with passage downstream and a doubling of mean values for 2010 between Kilbricken and New Bridge (Table 12) which is upstream of Durrow. A sequential increase in nitrate levels with passage downstream is apparent in the Nore and the same is seen in the other rivers.

Nitrite and nitrate results for designated salmonid rivers in the south-east, which also support *M. margaritifera*, for 2010, 2011 and January-March 2012 (n=1238) showed mean concentrations of 0.011 mg/l NO_2 and 2.6 mg/l N with ranges of <0.002-0.058 NO_2 and 0.05-5.84 mg/l N. The lower Blackwater had mean values of 0.010 mg/l NO_2 and 2.9mg/l N, Nore 0.012 mg/l NO_2 and 2.3mg/l N, Slaney 0.011 mg/l NO_2 and 3.4 mg/l N and Aherlow 0.010 mg/l NO_2 and 2.1 mg/l N (Table 13). These are all at the guideline threshold of 0.01 mg/l NO_2 in the parent Freshwater Fish Directive and would exceed the <1.7 mg/l N value suggested for protection of Irish pearl mussels. A sequential increase in nitrate levels with passage downstream is apparent in the Nore and the other rivers. Of all the study sites only eight, two in the Nore, Multeen and Thonoge and one each in the Derreen and Mountain rivers, would be regarded as satisfactory for pearl-mussel populations with respect of nitrite+nitrate-concentration conditions of <1.7 mg/l N. The mean nitrate value associated with the highest ecological quality (Q5) in Irish rivers is more than one-half lower again at 0.76 mg/l N and only one site would achieve that quality. With regard to the other nutrient phosphate the mean value for the four salmonid-designated rivers was collectively and individually 0.04 mg/l P which is above the objective for high status (≤0.025 mg/l P) in the Regulations giving effect to measures in the Water Framework Directive (WFD). Phosphate can affect mussels and juvenile salmonids indirectly by stimulating filamentous algal growths which can cover the river substratum and exert a smothering effect on the habitat.

Table 13 Nitrite, nitrate and phosphate measurements at four salmonid-designated rivers (January 2010-March 2012)

River	Range			Mean			Number of Samples (n)
	Nitrite (NO_2)	Nitrate (N)	Phosphate (P)	Nitrite (NO_2)	Nitrate (N)	Phosphate (P)	
Blackwater	<0.002-0.026	0.8-4.5	0.01-0.10	0.009	2.9	0.04	126
Nore	<0.002-0.058	0.05-5.8	0.01-0.48	0.012	2.3	0.04	613
Slaney	<0.002-0.043	0.9-5.6	0.01-0.14	0.011	3.4	0.04	332
Aherlow	<0.002-0.045	0.9-3.4	0.01-0.11	0.010	2.1	0.04	167

Of 31 river sites, for which data were available in 2010, in the study, only four, the Nore at Kilbricken, Mountain in Borris, Multeen at Augnagross and Thonoge at Ballyboley, would be regarded as satisfactory for pearl-mussel populations with respect of nitrite+nitrate-concentration conditions of <1.7 mg/l N (Table 14). Other sites, in the Derreen (Rathnagrew), Multeen (Ballygriffin), Nore (Poorman's Bridge) and Thonoge (Tubbrid) would marginally fail on this water quality criterion.

None of the sites would fail to comply with the nitrite limit of 0.05 mg/l NO_2 in the Irish Salmonid Regulations in 2010 although some would equal or marginally exceed the guideline value of 0.01 mg/l NO_2 in the parent directive – the Freshwater Fish Directive.

In the Box: Pearl Mussel Requirements (See Discussion), the physical and chemical habitat preferences are summarized. For the median nitrate value of <125 µg/l (0.125 mg/l), proposed for Scandinavian waters, it is not clear whether this is as NO_3 or N but would appear extremely low for British or Irish rivers. For British waters Skinner *et al*. (2003) have proposed nitrate and phosphate levels of <1.0 mg/l and <0.03 mg/l respectively. In relation to Irish populations Moorkens (2000) has proposed mandatory concentrations of <1.7 mg/l N and <0.06 mg/l P respectively for river nitrate and phosphate both of which would appear too lenient. The mean nitrate value associated with the highest ecological quality (Q5) in Irish rivers is more than one-half lower again at 0.76 mg/l N.

In a study of the pearl-mussel river Esk, in north-east England, land cover was found to be a key driver of nitrate concentration where high upstream percentage of improved pasture resulted in high nitrate concentration while high upstream percentage of moorland resulted in low nitrate concentration (Balmford, 2011). In south-east Ireland, where the current study was centred, a positive correlation between river nitrate levels and the proportion of ploughed land (tillage) in their catchments has been shown (Neill, 1989).

An increase in nitrate and phosphate and decrease in colour from upstream to downstream locations is apparent in the Nore. For the River Nore locations shown in Table 15 a comparison with median values[5] of 30 years previously (Flanagan and Neill, 1981) shows a slight reduction for phosphate and an increase for nitrate.

Only the Nore site at Poorman's Bridge would tentatively pass the requirements for nutrients (phosphate and nitrate). It would appear that nitrate levels, particularly in the south-east, are a compromising factor for pearl-mussel population.

5 For strict comparison the median is used for both years as this was the summary statistic employed in 1980.

Table 14 Mean values for nitrite and nitrate in 2010, where available, at study sites in core rivers (above) and non-core rivers (below)

River	Location	Nitrite (mg/l NO_2)	Nitrate (mg/l N)
Nore	Kilbricken	0.009	1.2
	Poorman's Bridge	0.011	1.7
	Waterloo Bridge	0.016	2.0
	Watercastle Bridge	0.015	2.1
	New Bridge	0.013	2.2
	Tallyho Bridge	0.015	2.7
	Ballyragget	0.016	2.8
	Threecastles	0.016	3.0
Blackwater	Ballyduff Bridge	0.011	3.2
	Lismore Bridge	0.011	3.2
	2 km d/s Lismore	0.011	3.2
Derreen (Slaney)	Rathnagrew	0.003	1.7
	Knockeen	0.012	3.8
	Rathglass	0.010	4.2
Mountain (Barrow)	Borris	0.001	1.5
Ballymurphy (Barrow)	Ballyroughan Little	0.007	3.9
Aughavaud (Barrow)	St. Mullin's	0.004	3.4
Clodiagh (Suir)	Clonea Bridge	0.004	2.1
	Lowry's Bridge	0.006	2.5
	Portlaw Bridge	0.005	2.4
Licky*	Licky Bridge	0.001	3.1

*2006 data

River	Locations	Nitrite (mg/l NO_2)	Nitrate (mg/l N)
Slaney	Rathmore Bridge	0.009	2.9
	Kilcarry Bridge	0.014	3.7
Multeen	Aughnagross Bridge	0.007	1.2
	Ballygriffin Bridge	0.007	1.7
Aherlow	Old Cappa Bridge	0.006	1.8
	Kilardry Bridge	0.009	2.1
Suir	Marlfield	0.006	2.4
Tar	Upstream Tar Bridge	0.003	2.1
Thonoge	Ballyboley	0.012	<0.1
	Tubbrid	0.004	1.7

Table 15 Median values for phosphate (MRP) and nitrate (TON) in the River Nore in 1980 and 2010

Location	Phosphate (mg/l P)		Nitrate (mg/l N)	
	1980	2010	1980	2010
Poorman's Bridge	0.04	0.02	1.3	1.5
New Bridge	0.05	0.04	1.4	2.4

OXYGEN STUDIES

Dissolved oxygen studies were carried out at the above locations on the River Nore as well as at 38 sites on 14 rivers overall. The Kilbricken and Ballyragget sites on the River Nore were tested for dissolved oxygen and redox potential over the four-week period of the sediment accumulation trial. Table 16 and Table 17 show the mean dissolved oxygen measurements taken on an average of five times over the duration of the project.

Table 16 Average Dissolved Oxygen (% saturation) results at Kilbricken (above) and Ballyragget (below) taken over the sample period

Dissolved Oxygen (%)	Week 1	Week 2	Week 3	Week 4
Free-flowing Water	100	101	100	101
Interstitial Water	93	90	95	94

Dissolved Oxygen (%)	Week 1	Week 2	Week 3	Week 4
Free-flowing Water	101	101	100	101
Interstitial Water	96	90	91	89

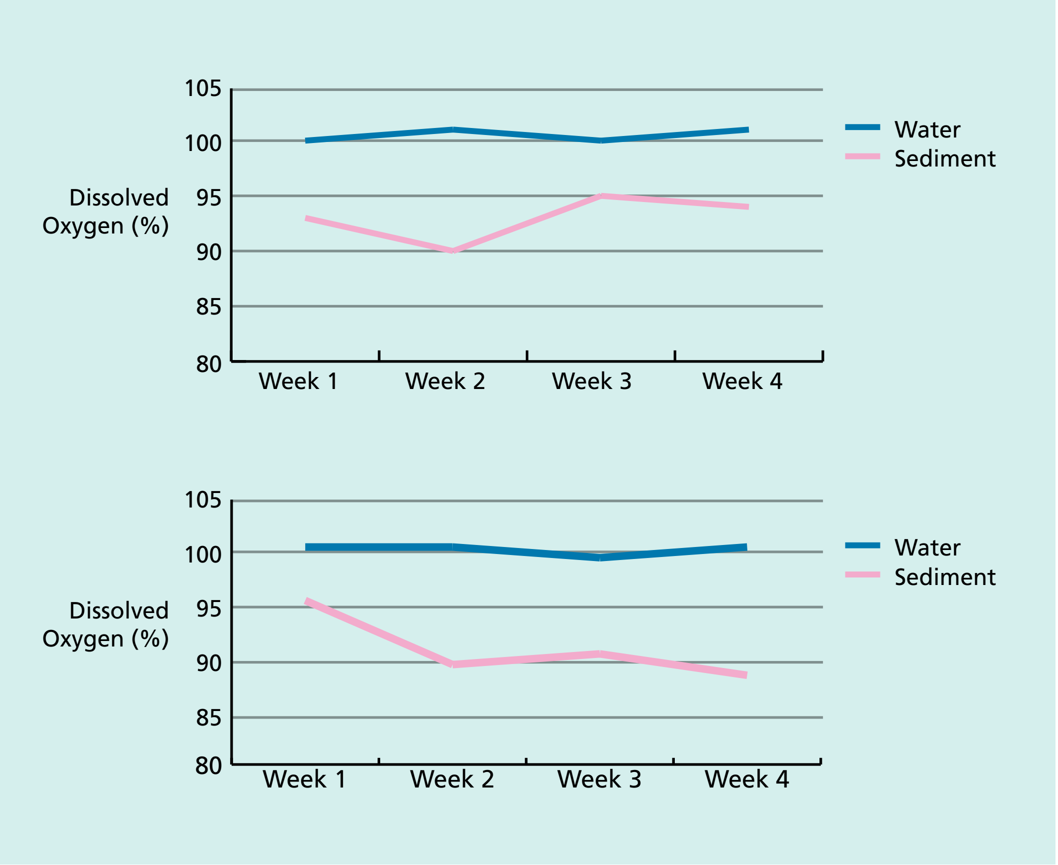


Figure 16 Showing average percentage oxygen saturation results for water and sediment at Kilbricken (above) and Ballyragget (below) over four-week period

Table 17 shows the mean dissolved oxygen concentration for each of the project core Rivers protected under the Freshwater Pearl Mussel Regulations. The mean was calculated from the results of all the readings at each of the river sites. The Derreen showed the highest level of dissolved oxygen, supersaturation, indicating some euthrophication. The Munster Blackwater shows the lowest level of interstitial water at 3cm. The interstitial water results at 10cm in field conditions are only included sparingly in this report as there were much less readings taken at this depth because of problem inserting the probe into the deeper riverbed substrate. For example, the average dissolved oxygen saturation, of six samples, at this depth in the Nore was 85 per cent where those for free-flowing water and 3cm were respectively 105 and 92 per cent (Table 17) for 36 measurements.

The results of dissolved oxygen from the non-core rivers are generally lower. The Aherlow has the highest level of dissolved oxygen in the free-flowing and interstitial water while the Multeen has the lowest level of interstitial dissolved oxygen at 69 per cent. The three sites on the Multeen River (Black Bridge, Aughnagross and Ballygriffen Bridge), which were silted, had average oxygen readings of 104, 69 and 57 per cent saturation in water column, 3cm and 10cm depth.

Table 17 Showing the mean dissolved oxygen (% saturation) in free-flowing water and in the interstitial water at 3cm for the core river sites (above) and non-core river sites (below)

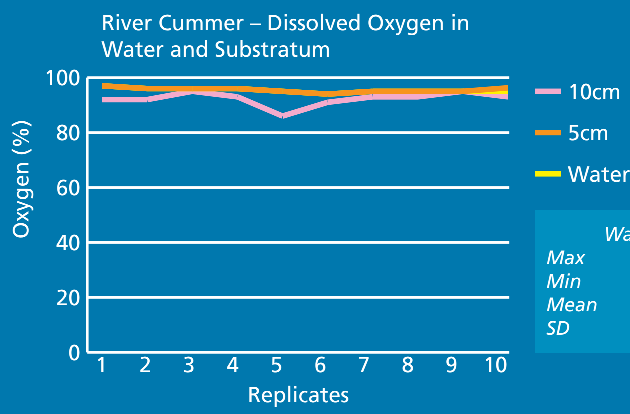
Dissolved Oxygen (%)	Free-flowing water	Interstitial 3cm
Nore	105	92
Blackwater	106	81
Derreen	115	101
Mountain	104	96
Ballymurphy	106	95
Aughavaud	105	99
Clodiagh	106	98
Licky	106	87

Dissolved Oxygen (%)	Free-flowing water	Interstitial 3cm
Slaney	108	84
Multeen	104	69
Suir	99	94
Tar	106	82
Thonoge	100	83
Aherlow	113	101

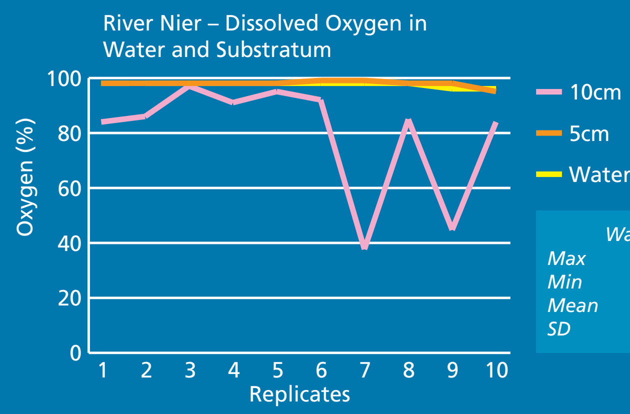
Figure 17 Showing measurement of dissolved oxygen (Detail – probe at 5cm in river substratum)

DISSOLVED OXYGEN IN RIVER GRAVELS

In July and September 2011 respectively, dissolved oxygen was measured in the water column (Conductivity 206 and 106 µS/cm; pH 8.12 and 7.80; Temperature 14.7˚C and 12.3˚C) at a site in each of the rivers Cummer (Cork) and Nier (Waterford). Ten replicate sample measurements of dissolved oxygen (water column, five and ten centimetres in the substratum) were taken across the river width.



	Water (%)	5cm (%)	10cm (%)
Max	97	97	95
Min	94	94	86
Mean	95.5	95.4	92.3
SD	0.85	0.84	2.54

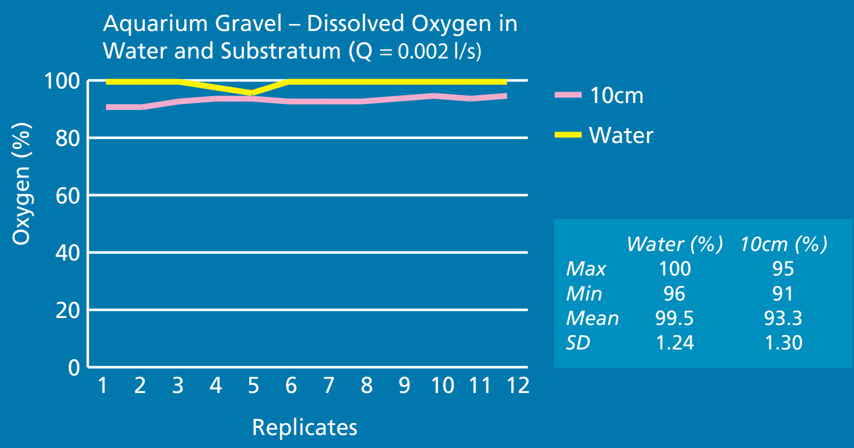


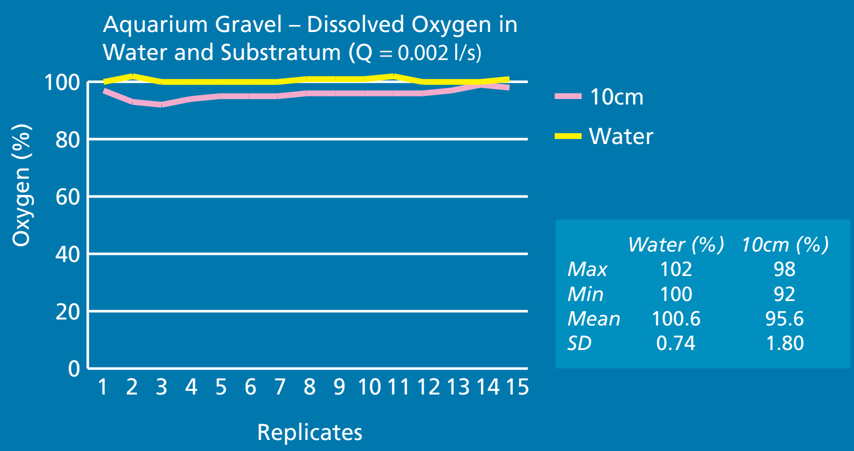
	Water (%)	5cm (%)	10cm (%)
Max	99	98	97
Min	95	96	38
Mean	97.9	97.6	92.3
SD	1.10	0.84	20.71

Both sites can be described as Glide areas and the substrata were predominantly gravel with sand but the Nier location also contained some silt. When the electrode was being withdrawn from two of the 10cm-replicate points in the Nier a plume of silt was released where the lowest oxygen levels were recorded; the relatively large standard deviation (SD) reflects this variation. The ecological water quality at the Cummer site was High and Good in the Nier.

DISSOLVED OXYGEN IN AQUARIUM GRAVELS

The luminescent dissolved oxygen meter was tested under laboratory conditions to check for reproducibility and repeatability using clean aquarium gravels. Tap water (Conductivity 530 µS/cm; pH 7.40) was trickled through gravel in a one-litre beaker, at two simulated flow regimes, volumetric flow rate (Q) = 0.002 and 0.013 l/S, and the dissolved oxygen was measured (water column and ten centimetres in the substratum) over time.





Both trials showed a consistent steady dissolved oxygen saturation reading in the water and substratum over time as would be expected in well-aerated water and clean aquarium gravels. The oxygen levels recorded in both water and gravels were greater when the volumetric flow was increased with mean differences of 1.1% and 2.3% respectively. In simulated deoxygenated conditions, using sodium sulphite (Na_2SO_3), the lowest saturation value recorded was 1.4% in both water and gravel.

REDOX STUDIES

Redox studies were carried out at most of the same locations on the River Nore as well as at sites on nine rivers overall. Table 18 and Table 19 show the mean redox measurements taken 4-5 times in total over the sample period.

Table 18 Average redox potential (mV) results at Kilbricken (above) and Ballyragget (below) taken over the sample period

Redox Potential (mV)	Week 1	Week 2	Week 3	Week 4
Free-flowing Water	214	258	256	253
Interstitial Water	101	90	66	79

Redox Potential (mV)	Week 1	Week 2	Week 3	Week 4
Free-flowing Water	221	234	216	270
Interstitial Water	-1	-6	-27	-50

Table 19 Showing the mean redox potential (mV) in free-flowing water and in the interstitial water at 3cm for the core river sites (above) and two of the non-core river sites (below)

Redox Potential (mV)	Free-flowing water	Interstitial 3cm
Nore	243	45
Blackwater	305	124
Derreen	258	135
Mountain	278	208
Aughavaud	281	177
Clodiagh	255	122
Licky	286	152

Redox Potential (mV)	Free-flowing water	Interstitial 3cm
Slaney	229	133
Multeen	228	-28

As can be seen from Tables 18 and 19 the interstitial measurements of redox potential with the exception of the Mountain River are below 200 which would indicate deoxygenation. However, direct measurements of dissolved oxygen did not substantiate this.

The pH and temperature at which measurements are made influence the redox potential of sediment samples[6]. In a classic study on marine sediments, it was found that as a very general rule, the pH of sediments increases with core depth and the redox decreases, or conditions become more alkaline and more reducing with core depth, i.e. the reducing capacity decreases with core depth (ZoBell, 1946). Curiously and conversely in the present study, conductivity, which can be used as an indirect measurement of alkalinity (EPA, 2000), was found to decrease with sediment depth when measured at a River Nore site (See Box: Redox Potential and River Gravels). Similarly in the laboratory much lower readings were measured in aquarium gravel than in the free water. However, this was subsequently ascribed to interference with the measurement of electrical resistance by 'clogging' of probe. It was concluded that the sediment was preventing contact between sensing elements and that such a meter was not suitable for measuring conductivity in sediment. In the laboratory no pH difference between water, 5cm and 10cm gravel depths, was apparent.

While a statistically significant inverse relationship (P=0.0042) was found for redox and conductivity in the hard-water Nore river this did not hold true when water results for all river sites (Figure 18) were computed (P=0.0894).

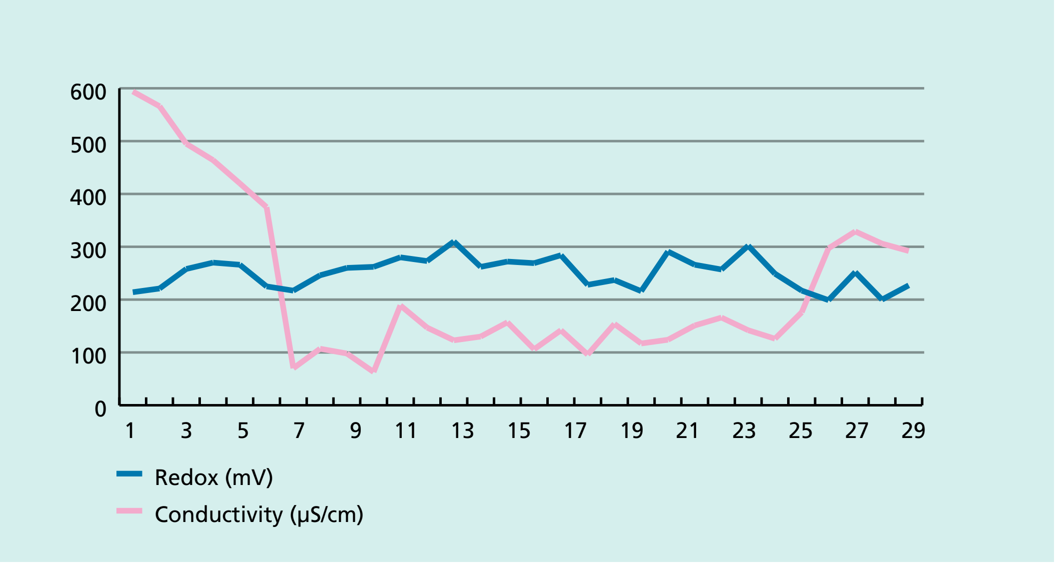


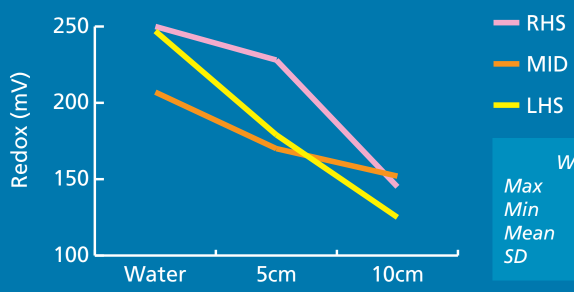
Figure 18 Showing measurements of redox potential (mV) and conductivity (µS/cm) in water at river sites

During the laboratory redox measurements it was found that results were very variable, e.g. rise in gravel readings from 181 to 250 mV in two-minute period and still rising reaching 265 mV after 10 minutes. As with all redox measurements for the project the initial reading, on insertion of probe, was used as there was significant drift with time. While an initial rise might be expected when probe is introduced, a further drift with time was evident. Servicing of the redox meter in 2011 improved its performance but there was still variability in results.

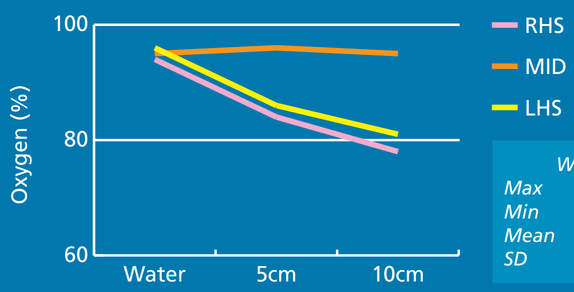
6 The redox potential is often corrected to pH 7 by adding 58 mV for every pH unit above 7 and by subtracting 58 mV for every pH unit below 7. The redox potential also depends marginally on temperature but decreases only by 60 mV from 0-30˚C (Søndergaard, 2010).

REDOX POTENTIAL IN RIVER GRAVELS

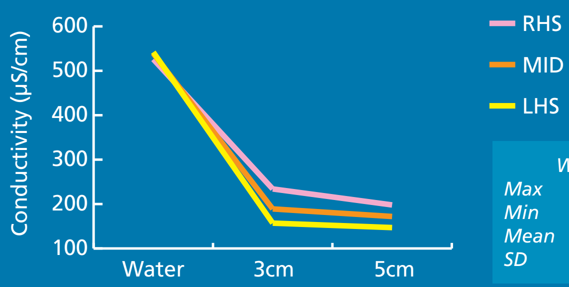
In March 2011 at a site in the river Nore (Laois) measurements of redox potential, dissolved oxygen and conductivity were taken (water column and in the substratum) across the river width. Readings were taken at left-hand side (LHS), middle (MID) and right-hand side (RHS) of the river channel in water and at 5 and 10cm in the substratum; except for conductivity which was taken at 3 and 5cm depth as difficulty in penetration into substratum due to wide protective cover on probe.



	Water (mV)	5cm (mV)	10cm (mV)
Max	250	228	152
Min	207	170	125
Mean	234.7	192.3	140.7
SD	24.01	31.21	14.01



	Water (mV)	5cm (mV)	10cm (mV)
Max	96	96	95
Min	94	84	84
Mean	95	88.7	92.3
SD	1.00	7.07	9.07

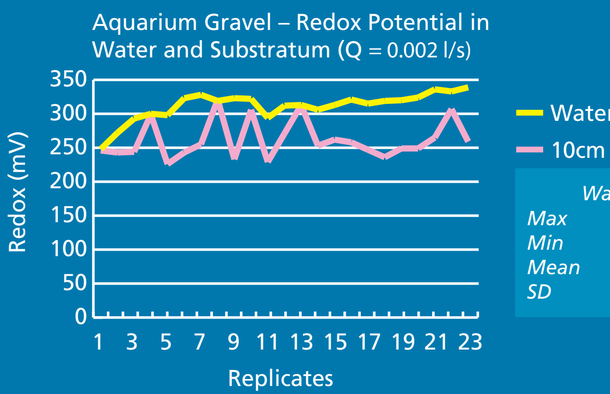


	Water (mV)	3cm (mV)	5cm (µS/cm)
Max	542	234	198
Min	526	157	147
Mean	535.7	193.3	172.3
SD	8.5	38.68	14.01

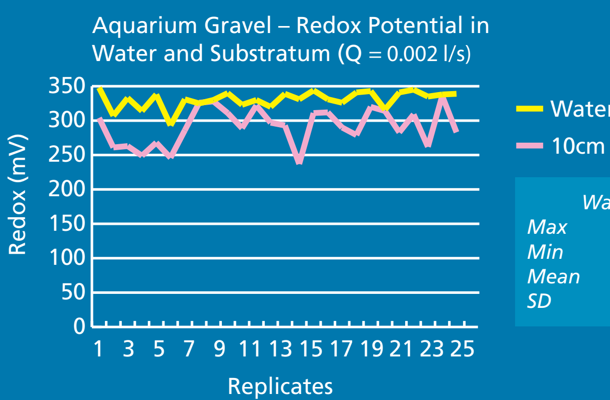
The site can be described as a Glide area and the substrata were predominantly gravel with sand. The results show that redox values were highest at RHS, as were conductivity, whereas oxygen levels were highest in midstream where flow was greater. These data gave the following correlations between parameters: Redox v Oxygen ($r^2 = 0.24$) Redox v Conductivity ($r^2 = 0.64$) Oxygen v Conductivity ($r^2 = 0.32$). The standard deviation (SD) for redox media, as well as for conductivity in substratum, was relatively large.

REDOX POTENTIAL IN AQUARIUM GRAVELS (1)

The redox meter was tested under laboratory conditions to check for reproducibility and repeatability using clean aquarium gravels. Tap water (Conductivity 530 µS/cm; pH 7.40; Temp. 18.0˚C) was trickled through gravel in a one-litre beaker, at two simulated flow regimes, volumetric flow rate (Q) = 0.002 and 0.016 l/S, and the redox potential was measured (water column and ten centimetres in the substratum) over time.



	Water (mV)	10cm (mV)
Max	339	321
Min	248	226
Mean	312	262
SD	20.78	28.14

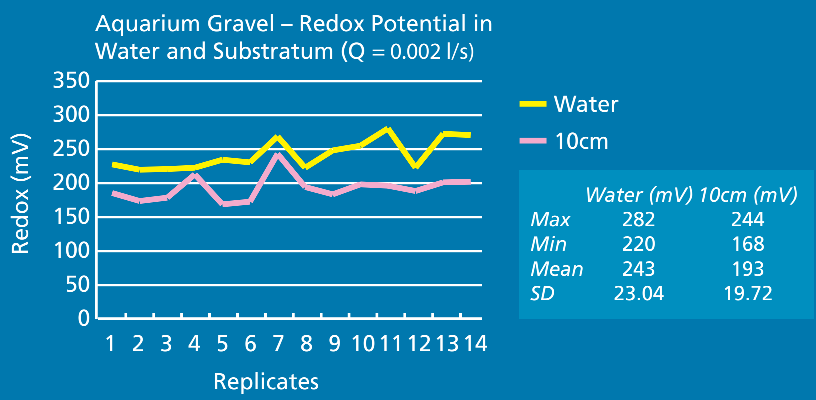


	Water (mV)	10cm (mV)
Max	348	336
Min	293	237
Mean	327	292
SD	12.97	27.41

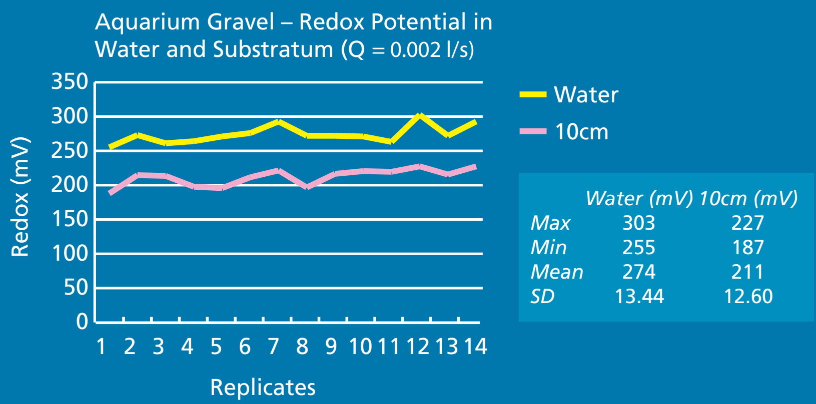
Both trials showed some variation in the water column and substratum and between these which is reflected in the standard deviation (SD) values. The redox levels recorded in both water and gravels were greater when the volumetric flow was increased with mean differences of 15 mV and 30 mV for water and substratum respectively. In simulated deoxygenated conditions, using sodium sulphite (Na_2SO_3), the lowest initial values recorded were -18 mV in water and -136 mV in gravel but readings varied over time.

REDOX POTENTIAL IN AQUARIUM GRAVELS (2)

The redox meter was again tested, after servicing, under laboratory conditions to check for reproducibility and repeatability using clean aquarium gravels in slightly higher water conductivity (626 μS/cm), pH (7.50) and temperature (20.0˚C). Again the water was trickled through gravel in a one-litre beaker, at two simulated flow regimes, volumetric flow rate (Q) = 0.002 and 0.016 l/S, and the redox potential was measured (water column and ten centimetres in the substratum) over time.



	Water (mV)	10cm (mV)
Max	282	244
Min	220	168
Mean	243	193
SD	23.04	19.72



	Water (mV)	10cm (mV)
Max	303	227
Min	255	187
Mean	274	211
SD	13.44	12.60

Both trials again showed some variation in the water column and substratum and between these which is reflected in the standard deviation (SD) values. The redox levels recorded in both water and gravels were greater when the volumetric flow was increased with mean differences of 31 mV and 18 mV for water and substratum respectively.

OXYGEN AND REDOX DATA

When the oxygen and redox measurements are graphed together it can be seen that there is large variation in the latter with negative values recorded in the Derreen and Nore (Figure 18). Regression analyses, for the locations in the six rivers shown, gave the following: Aughavaud r^2 = 0.0774; Ballymurphy r^2 = 0.3091; Mountain r^2 = 0.0932; Derreen r^2 = 0.1439; Blackwater r^2 = 0.0935; Nore r^2 = 0.1188. When negative readings of redox for the Nore were omitted from the analysis the coefficient of determination was reduced further, i.e. r^2 = 0.00005. However, there is a statistically significant relationship between oxygen and redox in two rivers: Ballymurphy (n=24) r = 0.5560 (P=0.0048); Mountain (n=60) r = 0.3052 (P=0.0177). No significant relationship was indicated for the other four rivers plotted in Figure 19 with the last of these not quite statistically significant: Auhghavaud r (n=23) = 0.2783 (P=0.1985); Blackwater r (n=12) = 0.3058 (P=0.3337); Derreen r (n=30) = 0.1151 (P=0.5447); Nore r (n=30) = 0.3446 (P=0.0622).

Laboratory experiments, using clean gravel and tap water at two simulated flow rates, were carried out to measure oxygen and redox potential in order to test equipment for reproducibility and repeatability. The results are summarized in the Boxes: Dissolved Oxygen in Aquarium Gravels and Redox Potential in Aquarium Gravels (1 and 2). For the laboratory oxygen trials the measurements showed consistent steady dissolved oxygen saturation values in the water and substratum over time. Mean loss in dissolved oxygen saturation between water and gravel was 6 and 5 per cent respectively for the initial and increased flow tests. In the redox trials some variation was recorded in water and substratum as well as between these. Mean loss in redox potential between water and gravel was 16 and 11 per cent respectively for the initial and increased flow experiments. The relatively high standard deviation values reflect the range in redox readings during the trials. When the values were corrected for temperature, e.g. +214 mV @ 18˚C, the differences reduced to 10 and 6.5 per cent respectively for redox loss.

In a further trial of redox readings when conductivity and temperature of the water were slightly higher and pH much the same, some differences in results emerged. Mean loss in redox potential between water and gravel was 21 and 23 per cent respectively for the initial and increased flow experiments. Again all readings were higher in the increased flow rate and as previously standard deviation of data was relatively high. Generally the redox potential was higher when conductivity was lower, e.g. tests of still water ranging in conductivity from 690 down to 550 µS/cm gave redox readings of 109 to 121 mV, but tended to equilibrate from around 500 down to 350 µS/cm.

In both experiments, oxygen and redox readings were higher in the increased flow trials as would, a priori, be expected in well-aerated water and clean aquarium gravel.

In a test of deoxygenated conditions, where oxygen was depleted using sodium sulphite (Na_2SO_3), the lowest recorded oxygen value was 1.4 per cent saturation in water and gravel (See Box: Dissolved Oxygen in Aquarium Gravels) which was steady over time. For redox potential the initial lowest value was – 136 mV (See Box: Redox Potential in Aquarium Gravels) but then readings varied over time, e.g. mean values of – 77 and – 150 mV for water and gravel could change to – 47 and – 117 mV respectively within minutes. The following day the deoxygenated water (1.4% saturation) gave redox readings ranging from – 34 to – 0.1 mV. During the laboratory tests under deoxygenated water conditions, pH increased to 8.85 while less expectedly conductivity decreased to 5 µS/cm but increased again, reaching 800-900 µS/cm, when sodium sulphite solution was diluted.

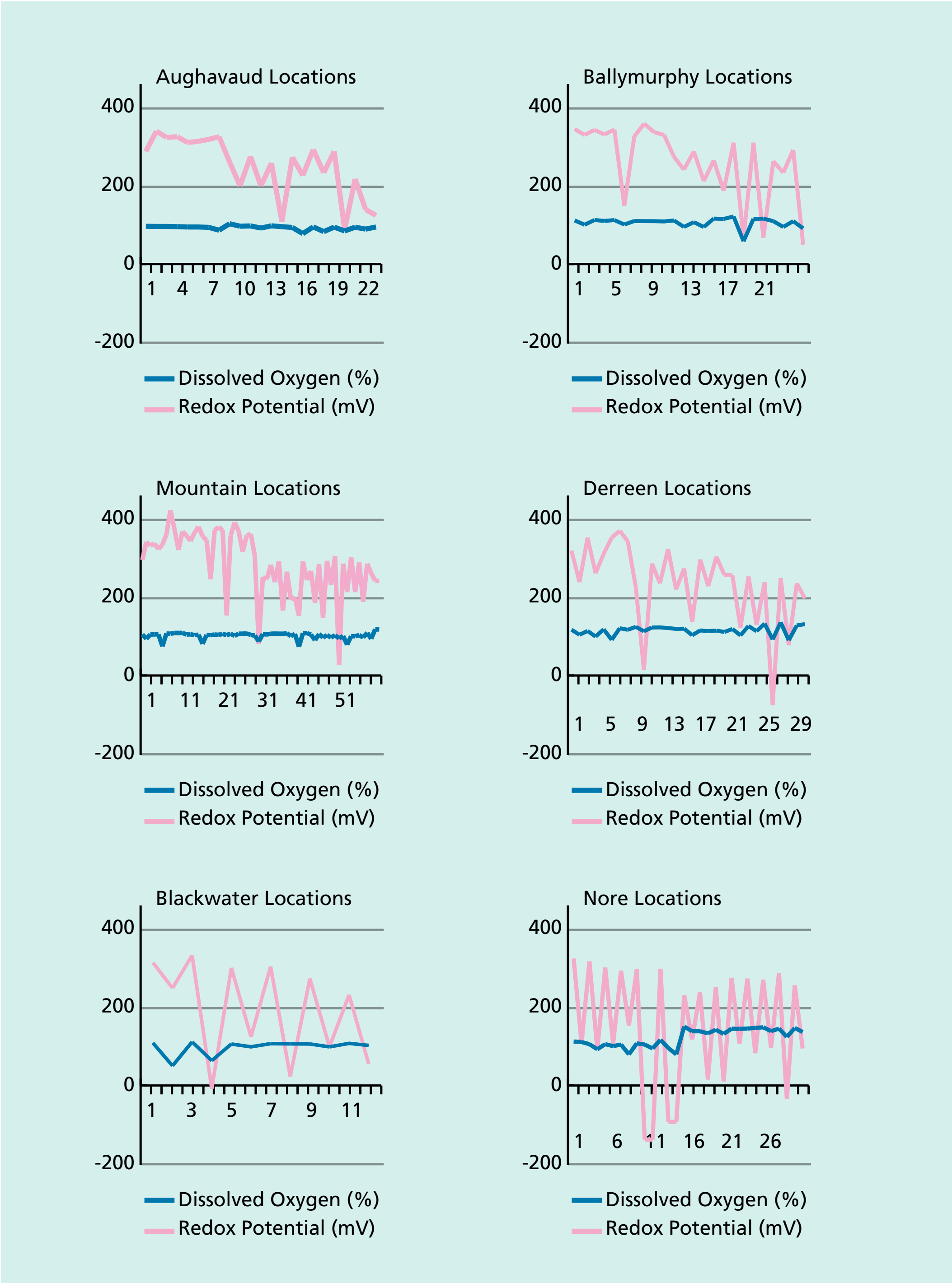


Figure 19 Redox (mV) and dissolved oxygen (% saturation) measurements in water at locations on six rivers. Some measurements were repeated at same sites, e.g. Mountain River

Based on differential results of dissolved oxygen and redox the sites would largely pass on the former and fail on the latter. In English rivers differentials between water and sediment of up to 60 per cent have been recorded in pearl-mussel rivers with 10 per cent in the cleanest (Killeen, 2010).

SALMONID FISH

A new ecological classification tool for fish in rivers (Fish Classification System – FCS2) is used, by IFI (Inland Fisheries Ireland), for Water Framework Directive monitoring where a section of river is assigned an 'ecological status' class of either High, Good, Moderate, Poor or Bad. Three stretches of the rivers covered in the present study, Nore, Multeen and Nier, have been classified respectively as Moderate, Good and High (IFI, 2010). The Nier is not a pearl-mussel river but is an important salmonid fishery with estimated densities of 0.66 and 0.05 per m^2 respectively for salmon (*Salmo salar*) and trout (*Salmo trutta*) compared with 0.001 and 0.01 per m^2 for a site in the Nore and 0.05 and 0.04 per m^2 for one in the Multeen (Central & Regional Fisheries Boards, 2009).

Unless salmonid fish are present, there can be no juvenile pearl mussels. As well as water and sediment quality the presence of these fish, in adequate numbers, is essential for pearl mussel survival. The salmon, in freshwater, is also protected under the EU Habitats Directive and both species' conservation should be considered together. The Nore is one of four rivers in the south-east designated as a salmonid river in accordance with the EU Freshwater Fish Directive (78/659/EEC).

During the project one of the authors accompanied Inland Fisheries Ireland (IFI) staff at some sites when catchment-wide electric-fishing (CWEF) surveys were being undertaken and data were kindly provided to the project subsequently.

This rapid semi-quantitative sampling technique is used primarily to assess the abundance of salmon fry (0+ fry) in riffle areas accessible to salmon throughout a given catchment and is undertaken in selected catchments nationally by IFI annually. From Table 20 it is can be seen that while all sites contained salmonids the number ranged between 0.08 and 0.75 per m^2.

Table 20 Salmonid fish (fry or parr) numbers at some of the pearl-mussel SAC sites during catchment-wide electric-fishing (CWEF) surveys carried out in July and August 2010: CWEF is a semi-quantitative timed (5-minute) electric-fishing sampling method. Missed fish are not included in catch number. Based on data supplied by Inland Fisheries Ireland

River	Location	Salmonid Fish			
		Salmon	Trout	(Missed)	Number/m^2
Nore	Kilbricken Bridge	34	1	(7)	0.46
	Poorman's Bridge	12	1	(6)	0.08
	Watercastle Bridge	20	1	(5)	0.22
	New Bridge	12	2	(3)	0.29
	Ballyragget Bridge	42	3	(13)	0.75
Mountain (Barrow)	Rosdelig Bridge	15	6	(4)	0.12
	Lacken Bridge	15	4	(3)	0.15
	Kilcoltrim Bridge	17	3	(3)	0.13
Ballymurphy (Barrow)	Earl's Bridge	6	12	(3)	0.20
Aughavaud (Barrow)	St. Mullin's	5	7	(3)	0.08

One site in the Nore as well as the one in the Aughavaud had lower numbers than the others and two further sites in the Nore had higher numbers present. It is interesting to note that the ratio of salmon to trout, with the exceptions of the Ballymurphy and Aughavaud sites, was at least 2:1 and as high as 34:1. The dominance ratio is reversed in the Ballymurphy and Aughavaud sites with trout 2:1 and 1.4:1 respectively.

Table 21 Mean values for physico-chemical parameters at some of the pearl-mussel SAC sites for which salmonid fish data are also available

River	Location	Redox (mV)	Oxygen (%)	Nitrate (mg/l N)	Phosphate (mg/l P)	Colour (Hazen)
		Water/Sediment				
Nore	Poorman's Bridge	273/48	109/97	1.7	0.02	70
	New Bridge	260/ -18	109/104	2.4	0.03	44
Mountain (Barrow)	Rosdelig Bridge	304/276	107/99	1.3	0.03	74
Ballymurphy (Barrow)	Earl's Bridge*	319/186	111/97	3.9	0.05	32
Aughavaud (Barrow)	St. Mullin's	298/222	106/98	3.4	0.04	34

*Water samples taken 2 km further downstream

In Table 21 physico-chemical data for some of the pearl-mussel SAC sites for which salmonid fish data were also available are given. These show good sediment oxygen conditions, with variable redox potential readings, in all but the nitrate and phosphate levels would indicate some nutrient enrichment.

SILTATION OBSERVATIONS

No physico-chemical objectives or guidelines have been set out as measures for protection of the mussel's habitat in Ireland but rather Ecological Quality Objectives for the elements macroinvertebrates, filamentous algae, phytobenthos, macrophytes and siltation, have been set down in legislation (European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009). These elements which include siltation are shown in Table 22.

Table 22 Ecological Quality Objectives for Freshwater Pearl Mussel Habitat

Element	Objective	Notes
Macroinvertebrates	EQR ≥0.90	High status
Filamentous algae (Macroalgae)	Absent or Trace (<5%)	Any filamentous algae should be wispy and ephemeral and never form mats
Phytobenthos (Diatoms)	EQR ≥0.93	High status
Macrophytes – rooted higher plants	Absent or Trace (<5%)	Rooted macrophytes should be absent or rare within the mussel habitat
Siltation	No artificially elevated levels of siltation	No plumes of silt when substratum is disturbed

Sand will, when present, be naturally released when the substratum is disturbed and it is important to distinguish this fraction from silt. The whole question of visual assessment of the degree of siltation is down to expert judgement and there is certainly a good argument for refining methodology to make results more scientifically robust.

Table 23a and Table 23b show the degree of siltation recorded at the study sites. In most situations, siltation was only observed at the sides of rivers (LHS or RHS) where current speed is generally slower and deposition is natural. For example, the Ballymurphy at Earl's Bridge on RHS and the Nore at Maddockstown on LHS had heavy siltation.

Very few sites exhibited excessive siltation but the Multeen, an otherwise clean river, did with one of the sites heavily silted at RHS. Similarly, the Licky on RHS at its uppermost location exhibited heavy siltation. Siltation can vary at sites over time by dissipation or by accumulating further depending on floods and other events. This is illustrated from observations in the national biological monitoring programme (M. McGarrigle, personal communication) for some of these sites. For example, Aughnagross Bridge on the Multeen River was clean in September 2008, clean with moderate siltation at LHS in June 2010 (this study) and silted to a slight degree in August 2011. Of the ten sites surveyed on the Multeen, in the national biological monitoring programme, four were clean, five were slightly silted and one was moderately silted. Similarly, Kilbricken on the River Nore was rated clean to slightly silted in September 2007, clean in June 2010 (this study) and clean again in October 2010 while Ballyragget was slightly silted in July 2010 and again some three months later in October 2010. Of the seven sites common to this study and the national biological programme, five were clean, one was slightly silted and another was slightly to moderately silted in October 2010 in the latter.

Table 23a Rivers protected under the Freshwater Pearl Mussel Regulations with observations on degree of siltation at sites

River	Location	Degree of siltation (Clean-Slight-Moderate-Heavy)
Nore	Kilbricken Bridge	Clean
	Poorman's Bridge	Moderate at RHS
	Waterloo Bridge	Clean
	Watercastle	Clean
	New Bridge	Moderate at RHS
	Tallyho Bridge	Moderate at RHS
	Ballyragget	Slight
	Knapton Bridge	Clean
	Threecastles	Moderate at RHS
	Maddockstown	Heavy at LHS
	Inchbeg	Clean
	Warrington	Slight-Moderate
Blackwater	Ballyduff Bridge	Clean
	Lismore Bridge	Moderate at LHS
	2 km d/s Lismore	Heavy at RHS
	u/s Cappoquin	Heavy at RHS
Derreen (Slaney)	Ballykilmurray	Clean
	Knockeen	Clean
	Rathnagrew	Clean
	Rathglass	Clean
	Ballyduff	Clean
Mountain (Barrow)	Upstream of Viaduct	Clean
	Kiledmond	Clean
	Rossdelig	Clean
	Owlbeg	Clean
	Lacken Bridge	Clean
	Borris	Slight-Moderate
	u/s Barrow	Clean
Ballymurphy (Barrow)	Ballyroughan Little	Clean
	Earl's Bridge	Heavy at RHS
	Cullentragh	Clean
Aughavaud (Barrow)	Turra Bridge	Clean
	St. Mullin's	Clean
Clodiagh (Suir)	Clonea Bridge	Moderate at LHS
	Glenstown Bridge	Clean
	Greens Bridge	Slight
	Lowry's Bridge	Slight
	Portlaw Bridge	Clean
Licky	Carrigeen	Heavy at RHS
	Grallagh	Moderate at LHS and RHS
	Licky Bridge	Moderate at LHS

Table 23b Other rivers with observations on degree of siltation at sites

River	Location	Degree of siltation (Clean-Slight-Moderate-Heavy)
Slaney	Aghade Bridge	Clean
	Rathmore Bridge	Clean
	Motaboher Bridge	Clean
	Kilcarry Bridge	Clean
Multeen	Black Bridge	Heavy at RHS
	Aughnagross	Moderate at LHS
	Ballygriffin	Moderate at LHS
Aherlow	Old Cappa Bridge	Clean
	Kilardry Bridge	Clean
Suir	Marlfield	Clean
Tar	Kilganny Bridge	Clean
	Goat's Bridge	Clean
	Upstream Tar Bridge	Clean
Thonoge	Ballyboley	Clean
	Tubbrid	Clean

Of the 56 sites covered in the present study, 35 were clean, 16 were clean with some degree of siltation at margins, three were slightly silted and two others were slightly to moderately affected (Table 23a and Table 23b).

DISCUSSION

Apart from depending on salmonid fish to complete their life-cycle, the exact requirements of pearl mussels are relatively unknown. However, water and sediment quality of their habitat appear to be decisive factors particularly for juvenile survival and are implicated in the decline of freshwater pearl-mussel populations throughout their range. Population declines have been caused by a number of factors, which include pearl-fishing, pollution, organic enrichment, siltation, river engineering and declining salmonid stocks (Langan *et al*., 2007). With the obvious exclusion of fish stocks and if poaching is substituted for pearl-fishing, then the same factors are also responsible for declining salmonid stocks.

Sediment load of river beds is often not only determined by areas adjacent to the river but can be influenced by sediments which eroded several kilometres away (Denic *et al*., 2010). Modelling of water and tillage erosion rates across Europe suggests that soil is being lost at a rate greater than it can be replenished by natural soil formation which can result in nutrient and sediment loss to waterways (Regan *et al*., 2012). Depending on rainfall, suspended solids loss from grazed grassland in Ireland can range between 45 and 90 kg per hectare (Tunney *et al*., 2007) which is relatively low compared to losses of several tonnes per hectare that can occur with severe soil erosion under tillage conditions (Morgan *et al*., 1998). Approximately 80 per cent of agricultural land is devoted to grass, 11 per cent to rough grazing and nine per cent to arable cereal and crop production (Regan *et al*., 2012).

Long-term data sets are needed to obtain reliable patterns of sediment retention in free flowing rivers which will require a well-established method for the quantification of sediment accumulation. For the purposes of the present study, sediment traps were used. There are four main types of samplers, all of which have a number of features in common, for suspended sediments: integrated samplers; instantaneous grab samplers; pump samplers and sediment traps (Ongley, 1996). A study in Sweden, using modified Whitlock-Vibert boxes, found that mean sedimentation of inorganic and organic fine material was significantly lower in streams with recent M. *margariitifera* recruitment than in those without recent recruitment, e.g. 2.4 ± 0.91 g/box and 9.9 ± 2.9 g/box inorganic and 0.31 ± 0.22 g/box and 0.93 ± 0.23 g/box organic for fines, i.e. <63µm size (Österling *et al*., 2010). The equivalent results from the present study, while not directly comparable when left for four weeks as opposed to three months, would equate to 1.59 g at Kilbricken and 1.53 g at Ballyragget for fines. There is a need for standardization of methodology so that various studies can be judiciously compared. All sediment trapping methods have their drawbacks. The use of solid walled buckets, such as deployed in the Dawros and Nore studies, are biased to deposition of fines from the water column and do not take into account the rate of deposition associated with inter-gravel flow which accounts for up to 25 per cent of all fines within a gravel substrate (Naden *et al*., 2003). However they are inexpensive and require no maintenance apart from the installation and extraction and as a proxy they show a good indication of the depositional rate of sediment particles over a set period of time (Hall, 2008). The mean estimates for sedimentation rates for fines (clay and silt) at the two Nore sites, with mean flows of 0.67 and 3.20 m^3/S, gave very similar results: 8.73 g/m^2/d at Kilbricken and 8.83 g/m^2/d at Ballyragget. These rates are higher than those estimated for the Dawros a spate river in Co Galway.

Traditionally, physical and visual methods have been used to quantify the volume of deposited fine sediment in rivers and streams. Recently, a biological metric for the assessment of fine sediment accumulation in UK rivers, using macroinvertebrate community response, has been developed (Extence *et al*., 2011). That proxy index (PSI – Proportion of Sediment-sensitive Invertebrates) lists the sensitivities of species and families of British benthic macroinvertebrates according to one of four Fine Sediment Sensitivity Ratings (FSSR), ranging from A (Highly sensitive) to D (Highly insensitive), where the two Irish riverine protected species, pearl mussel (*Margaritifera margaritifera*) and crayfish (*Austropotamobius pallipes*), are respectively designated A and B (Moderately sensitive). Uniquely in Ireland, so far as has been established, these two species can share the same river habitat such as the Nore and some others (Lucey, 2006).

No physico-chemical objectives or guidelines have been set out as measures for protection of the mussel's habitat in Ireland but instead ecological and siltation objectives have been set down in legislation (European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009). The latter objective is that there should be no artificially elevated levels of siltation which is assessed as 'no visible plumes of silt when substratum is disturbed'. Observations of siltation at the 56 sites covered in the present study showed that 35 were clean, 16 were clean with some degree of siltation at margins, three were slightly silted and two others were slightly to moderately affected. The whole question of visual assessment of the degree of siltation is down to expert judgement and there is certainly a good argument for refining methodology to make results more scientifically robust.

The measurement of suspended solids (SSC) is laborious with laboratory analyses required and hence further expense while, on the other hand, turbidity is easily measured. To overcome this, efforts have been made to extrapolate or estimate suspended solids concentrations (SSC) from turbidity values. An adequate relationship between turbidity measured in the field and SSC should be expected in most situations (Gippel, 1995). However, a turbidity/SSC relationship can be affected by water colour, from dissolved organic compounds which can absorb more light than inorganics (Malcolm, 1985). While it has been said that coloured dissolved organic matter is unlikely to alter the turbidity reading by more than 10 per cent (Gippel, 1995), the indications, from Figure 11, are that turbidity is strongly influenced by colour interference at Kilbricken in the River Nore.

Apart from the colour influence at Kilbricken, it can be reasoned, from this limited aspect of the study that turbidity would seem not to be a reliable guide to suspended solids concentrations in reasonably steady-state flow conditions in the River Nore. The initial trial in July, during which flow rates changed and an apparent strong correlation developed, would suggest that to establish such a relationship it would be necessary to carry out measurements for both parameters over a wide range of flow conditions. Statistically, a very weak relationship was found between turbidity and suspended solids in the River Nore.

It is often assumed that turbidity provides a direct measure of suspended sediment and that there is a formula or set of conversion factors with which SSC can be calculated from NTUs[7]. This is simply not the case and no such formulas exist (Downing, 2008). Each situation is different and a site would have to be calibrated to find the relationship. While turbidity was found to be an excellent indicator of suspended sediment concentration at the Ballea Bridge Lower gauging station on the River Owenabue, in Co Cork, with a coefficient of determination, r^2, of 0.967, the study highlighted the disproportionate impact that high intensity short duration flow events have on the transport of suspended sediment and emphasized the need for continuous monitoring of suspended sediment for accurately estimating suspended solids fluxes (Harrington & Harrington, 2011).

It can be concluded from the present study that if an association, between turbidity and suspended solids, does exist in the River Nore, then studies under a wide variety of hydrographical conditions would be necessary to establish any strong relationship indicated in the preliminary trial at Kilbricken. The amount of initial work necessary in establishing such a relationship largely nullifies any benefit although once a correlation, if one exists, is developed for a site then it would be useful as proxy for suspended solids concentration.

7 The degree of turbidity is not equal to the suspended solids concentration because turbidity is an expression of only one effect that the suspended solids have on the characteristics of water, i.e. the ability of light to penetrate through the water column. Thus, because the particle size and nature, e.g. organic v. inorganic, of the suspended solids affect the light scattering, different turbidity values can be measured for waters having the same suspended solids concentration (McKee and Wolf, 1963).

In a Swedish study, turbidity – as proxy for sedimentation – explained most of the difference between streams with (n=11) and without (n=13) recent pearl-mussel recruitment where mean turbidity was respectively 0.96 ± 0.14 NTU and 4.1 ± 1.4 NTU. From this a mean turbidity level of 1.0-1.9 NTU was extrapolated as a threshold value (Österling *et al*., 2010). Similarily, a turbidity limit of 1.0 FNU[8] in watercourses with reproducing pearl mussels has also been proposed (WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams* *http://ec.europa.eu/environment/life/themes/animalandplants/documents/fpm_streams.pdf*). Due to the high colour in some Irish *Margaritifera* rivers and streams, where resultant turbidity values of an order of magnitude higher than the suggested threshold are characteristic, this parameter would not seem to be appropriate. Because of the difficulty in obtaining necessary suspended solids information over a large range of hydrometric conditions the use of turbidity, which can be measured with hand-held meters or *in situ* recording devices, as a proxy is often employed. However, as found in the present study, the relationship between suspended solids and turbidity, if such exists, at a site can be difficult to establish.

At present there is no quality standard or objective for suspended and deposited solids under the Water Framework Directive (WFD). The Freshwater Fish Directive, which stipulates a guideline annual mean level of 25 mg/l, will be repealed by the WFD in 2013. A level of 30 mg/l of suspended solids has been given by Valovirta (1990) as the limit of tolerance by adult pearl mussels and reiterated, by Valovirta & Yrjänä (1997), that the maximum amount of fine material in the river water should be less than 25-30 mg/l. This level may not be critical if it occurs for a short time during flood but long-term levels of suspended solids should be much less while levels consistently above 10 mg/l should, according to Skinner *et al*. (2003), give cause for concern. The lower Blackwater, with a mean suspended solids loading of 16 mg/l, would therefore, on the latter criterion, be most at risk with respect to its mussel population. There is no evidence that concentrations of suspended solids less than 25 ppm have any harmful effects on fisheries (EIFAC, 1964). Thus, the suspended solids criterion for adult pearl mussels would appear to be of the same order as for freshwater fish.

Nitrate levels in rivers in the south-east have increased since sampling began in 1979, although concentrations have stabilised since the 1990s, while overall levels of phosphate have not increased significantly (Neill, 2010). Nationally, nitrate levels in Irish rivers exhibit an increase from west to east (Lucey, 2007). The increase in nitrate levels coincided with the intensification of agriculture in the 1970s. In a study of German rivers, mortality in adult pearl mussels showed a positive correlation with the concentration of nitrate in the water while phosphate, calcium and BOD (Biochemical Oxygen Demand) were correlated with decreasing survival and the establishment of the juveniles (Bauer, 1988).

Because of its toxicity to some aquatic organisms a maximum level of 2 mg/l N has been deemed appropriate for protecting the most sensitive freshwater species (Camargo *et al*., 2005). However, a maximum concentration of less than 1.7 mg/l N has been proposed as the quality requirement for sustainable Irish pearl-mussel water bodies (Moorkens, 2000). Acute toxicity testing of nitrite with the juvenile unionid *Lampsilis siliquoidea* found a 48-hour LC_{50} of 0.19 mg/l NO_2 indicating that it may be significantly toxic to young mussels (Myers-Kinzie, 1998). Yet a 96-hour LC_{50} of 922 mg/l N-NO_3 has been recorded for *Anodonta anatina* despite the probability of it and other unionid mussels' occurrence being significantly reduced in rivers with elevated nitrate levels (Douda, 2010). This latter species occurs in Ireland (Lucey, 1995) but its water quality tolerance, particularly regarding nutrient levels, would be much greater than that of *M. margaritifera*.

8 FNU = 1 FTU = 1 NTU

M. margaritifera has become extinct in the Barrow and Suir in the past 25-30 years and occurs in depleted numbers in the Nore, Slaney and Blackwater (See Figure 7 for location of rivers). The Nore population is under great threat of survival and not thought to be sustainable, due to enrichment, and there is a further upward trend in nitrate level apparent since 2003 (Lucey, 2007)

Comparison of nutrient levels with ecological Q-values shows a strong negative correlation in Irish rivers. Unimpacted sites with high ecological quality (Q5) correspond with mean Molybdate Reactive Phosphate (MRP) and Oxidised Nitrogen concentrations of 0.02 mg/l P and 0.76 mg/l N respectively or median values of 0.015 mg/l and 0.54 mg/l (McGarrigle *et al*., 1992). Phosphate is generally the limiting nutrient for plant growth in Irish rivers and raised levels can stimulate filamentous algae which can indirectly affect pearl mussels and salmonids. While the contrast between east and west rivers is not so marked for phosphate as for nitrate, some on the western seaboard also have very low concentrations (Lucey, 2009).

Nitrate levels rather than siltation would appear to be the problem for pearl mussels in most rivers in the south-east whereas the opposite would appear to be the case in the Multeen. Water quality criteria for nitrite of 0.08–0.35 mg/l NO_2 have been estimated for the protection of sensitive aquatic species (Camargo & Alonso, 2006) while a maximum level of 2 mg/l NO_3–N has been proposed for nitrate (Camargo *et al*. (2005).

From observations during the present study, siltation is most evident at lower flow regimes which normally corresponds with summer-autumn period in Ireland.

Redox potential is used as a proxy for sediment oxygen in Ireland and elsewhere in Europe but no overall relationship between the two parameters was found in the present study. Proponents of this methodology found redox potential at sites without pearl mussel recruitment differed markedly between the free-flowing water at the surface and at depths in the substrata whereas no differences were detectable at good quality sites. Redox potential (Eh) at the tested sites without recruitment in rivers throughout Europe differed markedly between the free-flowing water at the surface and at 5 and 10cm in the bed, whereas no differences were detectable at good quality sites. This was also true of electric conductivity and, to a lesser extent, pH. The stream bed at sites lacking pearl mussel recruitment had a more variable and higher penetration resistance, indicating clogging of the interstitial macropore system by the deposition of mud and compaction of the stream bed (Geist & Auerswald 2007). Irish pearl-mussel sites with juvenile recruitment were found to have no detectable differences between the redox potential (Eh) of the open water and the interstitial water at 5 or 10cm depth (North South Project 2, 2009). A measured redox potential of 300 mV in the bottom substrate indicates oxygen rich conditions and higher values indicate even better oxygen conditions (WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams*).

Although the redox potential is quick and very simple to measure, the results often vary considerably depending on the sampling and measurement technique used. Therefore, redox measurements are often used in relation to other environmental parameters and often regarded more as a relative rather than an exact measurement (Søndergaard, 2010).

The redox potential is a useful concept in explaining many chemical and biological phenomena related to water quality. However, the practical use of redox potential in natural ecosystems is fraught with difficulty. Commercially available devices for measuring redox potential do not all provide the same reading for the redox potential of well-oxygenated water or for a sample of reduced water or sediment. This makes it difficult to interpret redox potential measured in natural waters and sediments. Nevertheless, a decrease in redox potential below values obtained in oxygen-saturated water with a particular instrument indicates that reducing conditions are developing, and the reducing ability of the environment increases as the redox potential drops. Care must be taken not to introduce air or oxygenated water into the medium where redox potential is to be measured (Boyd, 2000).

Siltation is a major issue in the three English freshwater pearl mussel SAC rivers although there is a wide range in the severity of the problem. Measurements of redox potential were taken at a large number of locations within each river to provide information on siltation load (as a surrogate for oxygen) within the interstices of the substrate. The cleanest substrates in pearl mussel rivers showed a loss in redox potential of 20 per cent or less between the open water and the substrate at a depth of 5cm. In heavily silted parts of the rivers the loss in redox potential at 5cm depth exceeded 60 per cent (Killeen, 2010).

In the present study no such clear cut results were recorded and overall it has to be concluded that redox results were at times unreliable, erratic or inconsistent during field measurements.

Those employing redox methodology report that it is also possible to measure oxygen levels in the field directly but this is generally more difficult and is not recommended (WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams*).

As far back as the 1960s redox and oxygen were measured in soil 'at the same time'. However, while the platinum electrode could be inserted directly into the soil, a special soil-water chamber was required for the oxygen cathode (dropping mercury electrode) to function and measurements were restricted to the laboratory (Armstrong, 1967). In that study of waterlogged soils it was found that the ratio redox potential/oxygen diffusion was not constant: a slight increase in oxygen from zero was associated with a relatively large increase in oxidation-reduction potential; and as the oxygen level increased further, a relationship was still present, although the increase in potential per unit increase in oxygen diffusion was less. Thus, as oxygen increases the ratio becomes much lower, while variations become more obvious.

The major criticism of the use of optical measurement of dissolved oxygen was that the glass material became easily marked or scratched and eventually destroyed. The probe used here had a stainless steel guard (See Figure 2) and suffered no damage during the study as laboratory testing demonstrated. The one drawback with current design is lack of penetration into harder substrata which would benefit greatly if it was more slimline.

The small diameter of the tip of the redox platinum probe, at just over 1mm, had a distinct advantage over the oxygen and conductivity probes whose metal guards have diameters of respectively 16mm and 36mm. In most rivers in riffle and glide areas a hardpan was present preventing penetration to depths in substratum. The oxygen probe would benefit by being slimmer for penetration into such substrata. Compared with the redox electrode it did allow more ingress of water while being inserted to take measurements although settlement time was allowed in the case of oxygen measurements whereas redox readings were taken immediately on insertion. In the pearl-mussel river tested in Germany by Buddensiek *et al*. (1993), it was found that a firm plate of loamy soil and clay was reached at 10cm while in unionid mussel habitats this reached down to 20cm.

Physico-chemical parameters have also been listed among the habitat needs for the freshwater pearl mussel and those proposed for Scandinavian waters (WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams*). These as well as other habitat requirements are given in Box: Pearl Mussel Requirements.

Although early post-parasitic stages of *M. margaritifera* can survive for several weeks at low oxygen levels, a concentration of 6.0 mg/l (~65% saturation at 17.0˚C) has been reported as necessary to achieve normal growth in unionid mussels (Buddensiek *et al*., 1993).

PEARL MUSSEL REQUIREMENTS

The habitat requirements of pearl mussels are still imperfectly known. However, they do need high water quality, a stable riverbed consisting of a suitable material, good water flow in the hyporheic zone (interstitial sediments) and adequate availability of host fish. These mussels can be found in flowing and still waters.

The habitat for *Margaritifera margaritifera* must have a clean substrate free of inorganic silt, organic peat and detritus as these can block oxygen exchange to the buried juvenile mussels. In Scotland the physical microhabitat requirements of the mussel have been investigated in the River Kerry where adults and juveniles were found to have broadly the same habitat 'preferences'. However, adults were able to tolerate silty or muddy conditions for unknown lengths of time but juveniles were never found in this type of habitat. Optimal ranges for water depth and velocity were respectively 30-40cm and 0.25-0.75 m/S. In a study of 26 streams in seven European countries, sites with high stream bed quality, promoting pearl mussel populations with good juvenile recruitment, had coarser and better sorted substrata with significantly lower quantities of fines, and a higher manganese concentration in the fines, than poor quality sites. The stream bed at sites lacking pearl mussel recruitment had a more variable and higher penetration resistance indicating clogging of interstitial macropore system by the deposition of mud and compaction of the stream bed.

The scientific consensus is that *M. margaritifera* populations can only thrive in very high water quality conditions.

In Ireland minimum values for some water quality parameters were proposed for discussion in 2000 as follows:

pH <8.0, >6.3
Phosphates (Ortho-P) <0.06 mg/l P
Ammonia < 0.10 mg/l N
Conductivity <200 µS/cm
Nitrate (Ox. Nitrogen) <1.7 mg/l N
Biochemical Oxygen Demandl <3 mg/l O_2
Dissolved Oxygen >9 mg/l O_2

A mean concentration of ≤0.025 mg/l P for Molybdate Reactive Phosphorus (MRP) is now the objective for High status under the Water Framework Directive (WFD) Regulations.

The following water-quality guidelines have been proposed for Scandinavian pearl-mussel waters:

pH ≥ 6.2 (minimum)
Inorganic aluminium <30 µg/l (maximum)
Total phosphorus <5-15 µg/l (average)
Nitrate <125 µg/l (median)
Turbidity <1 FNU (average, spring flood)
Water colour <80 mg Pt/l (average, spring flood) – 1 Hazen unit is equivalent to 1 mg Pt/l
Water temperature <25°C (maximum)
Fine grain (<1mm) substrate <25 per cent (share of particles, maximum)
Redox potential >300 mV (corrected value)

No physico-chemical objectives or guidelines have been set out as measures for protection of the mussel's habitat in Ireland but rather Ecological Quality Objectives for the elements macroinvertebrates, filamentous algae, phytobenthos, macrophytes and siltation, have been set down in legislation.

Sources: Hastie *et al.*, 2000; Moorkens, 2000; Geist & Auerswald, 2007; WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams;* European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009; North South Project 2, 2009; European Communities Environmental Objectives (Surface Waters) Regulations 2009.

Methods have been used to sample interstitial water in mussel habitats (e.g. Buddensiek *et al*., 1990) but not for directly measuring dissolved oxygen. Given the instrumentation available, it is surprising that the present study appears to be the first time that the direct measurement of dissolved oxygen has been carried out in river substrata.

Silt can affect fish and mussels both directly, by clogging gills, of the adults and indirectly, by interfering with oxygen flow, in the egg or juvenile habitat. Declining salmonid populations in many rivers in England and Wales have been attributed to spawning gravel siltation and source fingerprinting has been undertaken to establish the source of fine interstitial sediment recovered from spawning gravels (Walling *et al*., 2003). Such an approach can provide useful information to distinguish between surface and channel/subsurface sources of the interstitial fines collected from the different rivers and thence programmes of measures can be put in place to redress the situation. The tolerance of incubating salmon embryos to spawning gravel sedimentation was examined under hatchery conditions and also in the River Bush, Co Antrim. In a laboratory assessment alevin survival was closely related to the level of fine material although no clear relationship was established between the level of fines and percentage survival in the wild (Cowx *et al*., 1998).

As juvenile mussels have a parasitic stage on salmonids before a free-living one, embedded within and then on the substratum, their population conservation should not be undertaken separately but rather in conjunction with these fish. In a study of two Scottish rivers the majority of encysted *M. margaritifera* glochidia (75-99%) appeared to be carried by 0+ salmon hosts while relatively small numbers appeared to be carried by trout hosts. Generally mussel abundance in those two rivers (Kerry and Moidart) appeared to be positively correlated with juvenile salmon abundance (Hastie & Young, 2003). However, in Scotland there are a number of other important *M. margaritifera* populations that are entirely trout-dependant as is the case in some Irish rivers.

Among the habitat needs for the freshwater pearl mussel, the number of juvenile salmonids has been given as ≥5 per 100 m^2, or 0.05 per m^2, for summer surveys (WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams*). Data from the rapid fish surveys in the present study, shown in Table 20, indicate that these 10 sites would fulfil the *M. margaritifera* requirement in relation to salmonid presence.

Pearl mussels would appear to be more susceptible to effects of siltation than their hosts – salmonid fish. A Swedish study concluded that recruitment failure of *M. margaritifera* appeared to be related to its own vulnerability to turbidity and sedimentation rather than to its host's response to this type of habitat degradation (Österling *et al*., 2010).

The 27 sub-basin catchments designated as SACs for pearl mussels in Ireland have been reviewed for fish status and it was concluded that there were probably enough available data for only one-third of these (North South Project 2, 2009). Although availability of salmonid host fish would appear not to be a limiting factor in Irish habitats it would nonetheless be judicious to have all pearl-mussel sites monitored for fish status.

Results of a study of Swedish streams suggest that it may be possible to evaluate the potential viability of mussel populations by measuring stream turbidity as a proxy for sedimentation (Österling *et al*., 2010). However, as was found in the present study this would not necessarily hold for those locations with naturally high water colour. Some indications of siltation were found in south-east Irish rivers but the more pressing issue would appear to be nutrient levels. Restoration measures to improve pearl mussel and salmonid fish habitats, if focused on reducing fine material transport into streams, would also help to reduce ingress of nutrients.

REFERENCES

1. Aldridge, K.T. & Ganf, G.G., 2003. Modification of sediment redox potential by three contrasting macrophytes: implications for phosphorus adsorption/desorption. *Marine and Freshwater Research* 54: 87-94.
2. American Public Health Association, American Water Works Association and Water Environment Federation, 2005. *Standard Methods for the Examination of Water & Wastewater.* 21st Edition. American Public Health Association, American Water Works Association and Water Environment Federation, Washington.
3. Armstrong, W., 1967. The relationship between oxidation-reduction potentials and oxygen-diffusion levels in some waterlogged organic soils. *Journal of Soil Science* 18: 27-34.
4. Balmford, D.S., 2011. Spatial and temporal water quality in the River Esk in relation to freshwater pearl mussels. Masters thesis, Durham University. Available at Durham E-Theses Online: http://etheses.dur.ac.uk/861/.
5. Bauer, G., 1988. Threats to the freshwater pearl mussels, *Margaritifera margaritifera* in Central Europe. *Biological Conservation* 45: 239-253
6. Bowden, C. 2011. *River Water Quality in Southeast Ireland 2010*. A Report Commissioned by the County Councils of Carlow, Kilkenny, Laois, North Tipperary, South Tipperary, Waterford & Wexford and by Waterford City Council. EPA, Kilkenny.
7. Bowden, C. 2012. *Integrated Water Quality Report: South East Ireland 2011*. EPA, Kilkenny.
8. Boyd, C.E., 2000. *Water Quality: An Introduction.* Kluwer Academic Publishers, Norwell.
9. Brunke, M., 1999. Colmation and depth filtration within streambeds: retention of particles in hyporheic interstices. *International Review of Hydrobiology* 84: 99-117.
10. Buddensiek V., Engel, H., Fleischhauer-Rössing, S., Olbrich, S. & Wächtler, K., 1990. Studies on the chemistry of interstitial water taken from defined horizons in the fine sediments of bivalve habitats in several northern German lowland waters I: sampling techniques. *Archiv für Hydrobiologie* 119: 55-64.
11. Buddensiek V., Engel, H., Fleischhauer-Rössing, S., & Wächtler, K., 1993. Studies on the chemistry of interstitial water taken from defined horizons in the fine sediments of bivalve habitats in several northern German lowland waters II: microhabitats of *Margaritifera margaritifera* L., *Unio crassus* Philipsson and *Unio tumidus* Philipsson. *Archiv für Hydrobiologie* 127: 151-166.
12. Camargo, J.A., Alonso, Á. & Salamanca, A. 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. *Chemosphere* 58: 1255-1267.
13. Camargo, J.A. & Alonso, Á., 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environment International* 32: 831–849.
14. CEC, (Council of the European Communities), 1978. Council Directive of 18 July 1978 on the quality of freshwaters needing protection or improvement in order to support fish life (78/659/EEC). *Official Journal of the European Communities*, No. L 222/1.
15. CEC (Council of the European Communities), 1992. Council Directive of the 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (92/43/EEC). *Official Journal of the European Communities*, No. L 06/35.
16. Central & Regional Fisheries Boards, 2009. *Sampling Fish for the Water Framework Directive – Rivers 2008: SRFB Rivers.* Central Fisheries Board, Dublin.
17. Chapman, D.W., 1988. Critical review of variables used to define effects of fines in redds of large salmonids. *Transactions of the American Fisheries Society* 117: 1-21.

18. Cowx, I.G., O'Grady, K.T., O'Connor, W.C.K. and Andrew, T.E., 1998. The effects of siltation on Atlantic salmon, Salmo salar L., embryos in the River Bush. *Fisheries Management and Ecology* 5: 393-401.

19. Denic, M., Strohmeier, P., Bruckner, G., Gum B. & Geist J., 2010. *Integrated Sediment Management in River Catchments.* Seminar on Conservation of Freshwater Pearl Mussels in the Ardennes (Luxembourg) 27 November 2009.

20. Department of the Environment, 1972. *Analysis of Raw, Potable and Waste Waters.* HMSO, London.

21. Douda, K., 2010. Effects of nitrate nitrogen pollution on Central European unionid bivalves revealed by distributional data and acute toxicity testing. *Aquatic Conservation: Marine and Freshwater Ecosystems* 20: 189–197.

22. Downing, J., 2008. *Comparison of Suspended Solids Concentration (SSC) and Turbidity*. Application Note 2Q-AA. Campbell Scientific, Inc., Logan.

23. EIFAC (European Inland Fisheries Advisory Commission), 1964. *Water Quality Criteria for European Freshwater Fish. Report on finely divided solids and inland fisheries.* EIFAC/T1. FAO, Rome.

24. EPA (Environmental Protection Agency), 2000. *The Hardness of Irish Drinking Waters: A Simple Guide Based on Conductivity Data.* EPA, Wexford.

25. EPA (Environmental Protection Agency), 2001. *Parameters of Water Quality.* EPA, Wexford.

26. European Commission, 2008. Commission Decision 2008/915/EC of 30 October 2008 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the member state monitoring system classifications as a result of the intercalibration exercise. *Official Journal of the European Union* L332:20-44.

27. European Communities (Quality of Salmonid Waters) Regulations 1988. S.I. No. 293/1988 — The European Communities (Quality of Salmonid Waters) Regulations 1988.

28. European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009. S.I. No. 296/2009 — The European Communities Environmental Objectives (Freshwater Pearl Mussel) Regulations 2009.

29. European Communities Environmental Objectives (Surface Waters) Regulations 2009. S.I. No. 272/2009 — The European Communities Environmental Objectives (Surface Waters) Regulations 2009.

30. Extence, C. A., Chadd, R. P., England, J., Dunbar, M. J., Wood, P. J. & Taylor, E. D., 2011. The assessment of fine sediment accumulation in rivers using macro-invertbrate community response. *River Research and Applications* 27: 10.1002/rra.1569.

31. Flanagan, P.J. & Neill, M., 1981. *River Water Quality in the South Eastern (R.D.O.) Region and in Tipperary (N.R.)*. A Report to the management Committee of the Kilkenny Regional Laboratory for the Year 1980. An Foras Forbartha, Dublin.

32. Geist, J & Auerswald, K, 2007. Physicochemical stream bed characteristics and recruitment of the freshwater pearl mussel (*Margaritifera margaritifera*). *Freshwater Biology* 52: 2299-2316.

33. Gippel, C.J. 1995. Potential of turbidity monitoring for measuring the transport of suspended solids in streams. *Hydrological Processes* 9: 83-97.

34. Hall, K., 2008. The status of the river Dawros as a habitat for the freshwater pearl mussel *Margaritifera margaritifera*. Unpublished Master's Thesis. Trinity College, Dublin.

35. Hall, K. & Allot, N., 2010. *Measuring sediment accumulation in Irish Pearl Mussel rivers: Preliminary results and proposed future methodology*. Poster Session at Seminar on Conservation of Freshwater Pearl Mussels in the Ardennes (Luxembourg) 27 November 2009.

36. Harrington, S. & Harrington, J., 2011.Turbidity based suspended sediment fluxes in the Owenabue catchment Ireland. In: I. Stengel, U. Bleimaan, B. Humm and P. Walsh (eds.) *Proceedings of the Collaborative European Research Conference 2011*, pp 101-111. Cork Institute of Technology, Cork.

37. Hastie, L.C., Boon, P.J. & Young, M.R., 2000. Physical microhabitat requirements of freshwater pearl mussels, *Margaritifera margaritifera* (L.). *Hydrobiologia* 429: 59-71.

38. Hastie, L.C. & Young, M.R., 2003. Conservation of the Freshwater *Pearl Mussel 2. Relationship with Salmonid*s. Conserving Natura 2000 Rivers Conservation Techniques Series No. 3. English Nature, Peterborough.

39. Heywood, M.J.T. & Walling, D.E., 2007. The sedimentation of salmonid spawning gravels in the Hamshire Avon catchment, UK: implications for the dissolved oxygen content of intragravel water and embryo survival. *Hydrological Processes* 21: 770-788.

40. Ingendahl, D., 2001. Dissolved oxygen concentration and emergence of sea trout fry from natural redds in tributaries of the River Rhine. *Journal of Fish Biology* 58: 325-341.

41. IFI (Inland Fisheries Ireland), 2010. *Sampling Fish for the Water Framework Directive. WFD Summary Report 2007-2009*. Inland Fisheries Ireland, Dublin.

42. Killeen, I.J., 2010. *Siltation in English FWPM SAC rivers*. Seminar on Conservation of Freshwater Pearl Mussels in the Ardennes (Luxembourg) 27 November 2009.

43. Kozerski, H.-P., 2002. Determination of areal sedimentation rates in rivers by using plate sediment trap measurements and flow velocity – settling flux relationship. *Water Research* 36: 2983-2990.

44. Kozerski, H.-P., 2003. Seston sedimentation in a lowland river (River Spree, Germany): their spatial and temporal variations and controlling factors. *Hydrobiologia* 494: 51-55.

45. Langan, S., Cooksley, S., Young, M., Stutter, M., Scougall, F., Dalziel, A., Feeney, I., Lilly, A. & Dunn S., 2007. *The management and conservation of the freshwater pearl mussel in Scottish catchments designated as Special Areas of Conservation or Sites of Special Scientific Interest.* Scottish Natural Heritage Commissioned Report No.249 (ROAME No. F05AC607).

46. Lucey, J., 1995. The distribution of *Anodonta cygnea* (L.) and *Anodonta anatina* (L.) (Mollusca: Bivalvia) in southern Irish rivers and streams with records from other areas. *Irish Naturalists' Journal 25*: 1-8.

47. Lucey, J., 2006. The pearl mussel, *Margaritifera margaritifera* (L.), in hard water in Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy* 106B: 143-153.

48. Lucey, J. (Comp.), 2007. *Water Quality in Ireland 2006: Key Indicators of the Aquatic Environment*. EPA, Wexford.

49. Lucey, J. (Comp.), 2009. *Water Quality in Ireland 2007-2008: Key Indicators of the Aquatic Environment.* EPA, Wexford.

50. Malcolm, R.L., 1985. Geochemistry of stream fulvic and humic substances. In: G.R. Aiken, D.M. Mcknight, R.L. Wershaw and P. MacCarthy (eds.) *Humic Substances in Soil, Sediment and Water*, pp 181–209. Wiley, New York.

51. McGarrigle. M.L., Lucey, J. & Clabby, K.J., 1992. Biological assessment of river water quality in Ireland. In: P.J. Newman, M.A. Piavaux and R.A. Sweeting (eds.) *River Water Quality – Ecological Assessment and Control,* pp 371-393. Commission of the European Communities, Luxembourg. EUR 14606 EN-FR.

52. McGarrigle. M., Lucey, J. & Ó Cinnéide, M. (eds.), 2010. *Water Quality in Ireland 2007-2009*. EPA, Wexford.

53. McKee, J.E. & Wolf. H.W. (eds.), 1963. *Water Quality Criteria*. State Water Quality Control Board, Sacramento.
54. Morgan, R.P.C., Quinton, J.N., Smith, R.E., Govers, G., Poesen, J.W.A., Auerswald, K., Chisci, G., Torri, D. & Styczen, M.E. 1998. The European Soil Erosion Model (EUROSEM): a dynamic approach for predicting sediment transport from fields and small catchments. *Earth Surface Processes and Landforms* 23: 527–44.
55. Moorkens, E.A., 2000. Conservation Management of the Freshwater Pearl Mussel *Margaritifera margaritifera*. Part 2: Water Quality Requirements. *Irish Wildlife Manuals*, No. 9.
56. Moorkens, E.A., 2010. Addressing the conservation and rehabilitation of *Margaritifera margaritifera* (L.) populations in the Republic of Ireland within the framework of the Habitats and Species Directive. *Journal of Conchology* 40: 339-350.
57. Myers-Kinzie, M.L., 1998. Factors affecting survival and recruitment of unionid mussels in small midwestern streams. Unpublished Doctoral thesis. Purdue University, Indiana.
58. Naden, P., Smith, B., Jarvie, H., Llewellyn, N., Matthiessen, P., Dawson, H., Scarlett, S. & Hornby, D., 2003. *Siltation in Rivers. A Review of Monitoring Techniques*. Conserving Natura 2000 Rivers Conservation Techniques Series No. 6. English Nature, Peterborough.
59. Neill, M. 1989. Nitrate concentrations in river waters in the south-east of Ireland and their relationships with agricultural practice. *Water Research* 23: 1339-1355.
60. Neill, 2010. *River Water Quality in Southeast Ireland, 2009*. A Report Commissioned by the County Councils of Carlow, Kilkenny, Laois, Tipperary (N&S), Waterford & Wexford and by Waterford City Council. EPA, Kilkenny.
61. Nordstrom, D.K. & Wilde, F.D., 1998. Reduction-oxidation potential (electrode method). In: F.D. Wilde, D.B. Radtke, J. Gibs and R.T. Iwatsubo (eds.) *National Field Manual for the Collection of Water-Quality Data.* U.S. Geological Survey Techniques in Water-Resources Investigations, Book 9: A6.5.
62. North South Project 2, 2009. *Monitoring Methods Report: Freshwater Pearl Mussel Sub-basin Plans*. Produced by NS 2, funded by DEHLG [Department of the Environment, Heritage & Local Government].
63. Ongley, E., 1996. Chapter 13 – Sediment Measurements. In: *Water Quality Monitoring – A Practical Guide to the Design and Implementation of Freshwater Quality Studies and Monitoring Programmes*. J. Bartram and R. Balance (eds.). United Nations Environment Programme and the World Health Organisation, London.
64. Österling, M.E., Arvidsson, B.L. & Greenberg, L.A., 2010. Habitat degradation and the decline of the threatened mussel *Margaritifera margaritifera*: influence of turbidity and sedimentation on the mussel and its host. *Journal of Applied Ecology* 47: 759-768.
65. Oxford Reference Shelf, 1994. *Oxford Reference Shelf CD – ROM*. Oxford University Press, Oxford.
66. Project LIFE-Nature, 2009. Restauration des populations de moules perlières en Ardennes - Life05Nat/L/000116. *Increased sedimentation, a widespread problem leading to degradation of freshwater habitats and communities.* International Seminar, Clervaux, Luxembourg, 25th–28th November, 2009. Fondation Hëllef fir d'Natur, Heinerscheid.
67. Regan, J.T., Fenton, O. & Healy, M.G., 2012. A review of phosphorus and sediment release from Irish tillage soils, the methods used to quantify losses and the current state of mitigation practice. *Biology and Environments: Proceedings of the Royal Irish Academy* 112B: 157-183.
68. SEPA (Scottish Environment Protection Agency), 2010. *Engineering in the Water Environment: Good Practice Guide. Sediment Management.* SEPA, Stirling.

69. Skinner, A., Young, M. & Hastie, L., 2003. *Ecology of the Freshwater Pearl Mussel*. Conserving Natura 2000 Rivers Ecology Series. No. 2 English Nature, Peterborough.

70. Søndergaard, M., 2010. Redox potential. In: G.E. Linkens (ed.) *Geochemistry of Inland Waters,* pp 549-556. Elsevier, Oxford.

71. Udden, J.A., 1914. Mechanical composition of clastic sediments. *Bulletin of the Geological Society of America* 25: 655-744.

72. Toner, P., Bowman, J., Clabby, K., Lucey, J., McGarrigle, M., Concannon, C., Clenaghan, C., Cunningham, P., Delaney, J., O'Boyle, S., MacCárthaigh, M., Craig, M. & Quinn, M., 2005. *Water Quality in Ireland 2001-2003*. EPA, Wexford.

73. Tunney, H., Kurz, I., Bourke, D., O'Reilly, C., Jeffrey, D., Dowding, P., Foy, B., Kilpatrick, D. & Haygarth, P., 2007. *Eutrophication from Agricultural Sources: the Impact of the Grazing Animal on Phosphorus, Nitrogen, Potassium and Suspended Solids Loss from Grazed Grassland – Field-Plot Study (2000-LS-2.1.2-M2) Final Report*. EPA, Wexford.

74. Valovirta, I., 1990. Conservation of *Margaritifera margaritifera* in Finland. In: *Invertebrates and their Conservation*. Colloquium of the Bern Convention. Strasbourg. Council of Europe. T-PVS (89) 34: 59–63.

75. Valovirta, I. & Yrjänä, T. 1997. Effects of restoration of salmon rivers on the mussel *Margaritifera. margaritifera* (L.) in Finland. In: *Convention on the conservation of European wildlife and natural habitats.* Colloquium on Conservation, Management and Restoration of Habitats for Invertebrates: Enhancing Biological Diversity. Killarney, 26-29 May 1996. Council of Europe. T-PVS (96) 51: 38-48.

76. Walling, D.E., Collins, A.L. & McMellin, G.K., 2003. A reconnaissance survey of the source of interstitial fine sediment recovered from salmonid spawning gravels in England and Wales. *Hydobiologia* 497: 91-108.

77. Wentworth, C.K., 1922. A scale of grade and class terms for clastic sediments. *Journal of Geology* 30: 377-392.

78. Winkler, L.W., 1888. Die Bestimmung des im Wasser gelösten Sauerstoffes. *Berichte der deutschen chemischen Gesellschaft* 21: 2843-2854.

79. WWF Sweden – *Restoration of Freshwater Pearl Mussel Streams. (http://ec.europa.eu/environment/life/themes/animalandplants/documents/fpm_streams.pdf)*

80. YSI, 2009. ProODO *User Manual*. YSI Incorporated, Yellow Springs (OH).

81. Zobell, C.E., 1946. Studies on redox potential of marine sediments. B*ulletin of the American Association of Petroleum Geologists* 30: 477-513.

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.

Environmental Protection Agency
An Ghníomhaireacht um Chaomhnú Comhshaoil

Headquarters

Johnstown Castle Estate
County Wexford, Ireland

T +353 53 916 0600
F +353 53 916 0699
E info@epa.ie
LoCall 1890 33 55 99

www.epa.ie

Regional Inspectorate
McCumiskey House
Richview, Clonskeagh Road
Dublin 14, Ireland

T +353 1 268 0100
F +353 1 268 0199

Regional Inspectorate
Inniscarra, County Cork
Ireland

T +353 21 487 5540
F +353 21 487 5545

Regional Inspectorate
Seville Lodge, Callan Road
Kilkenny, Ireland

T +353 56 779 6700
F +353 56 779 6798

Regional Inspectorate
John Moore Road, Castlebar
County Mayo, Ireland

T +353 94 904 8400
F +353 94 902 1934

Regional Inspectorate
The Glen, Monaghan, Ireland

T +353 47 77600
F +353 47 84987

Regional Offices

Annabella, Mallow
Co. Cork, Ireland
T +353 22 43865

The Civic Centre
Church St., Athlone
Co. Westmeath, Ireland
T +353 906 475722

Mungret College
Mungret, Co. Limerick
Ireland
T +353 61 227900

11 Rosemount Lane
Letterkenny, Co. Donegal
Ireland
T +353 74 912 2274